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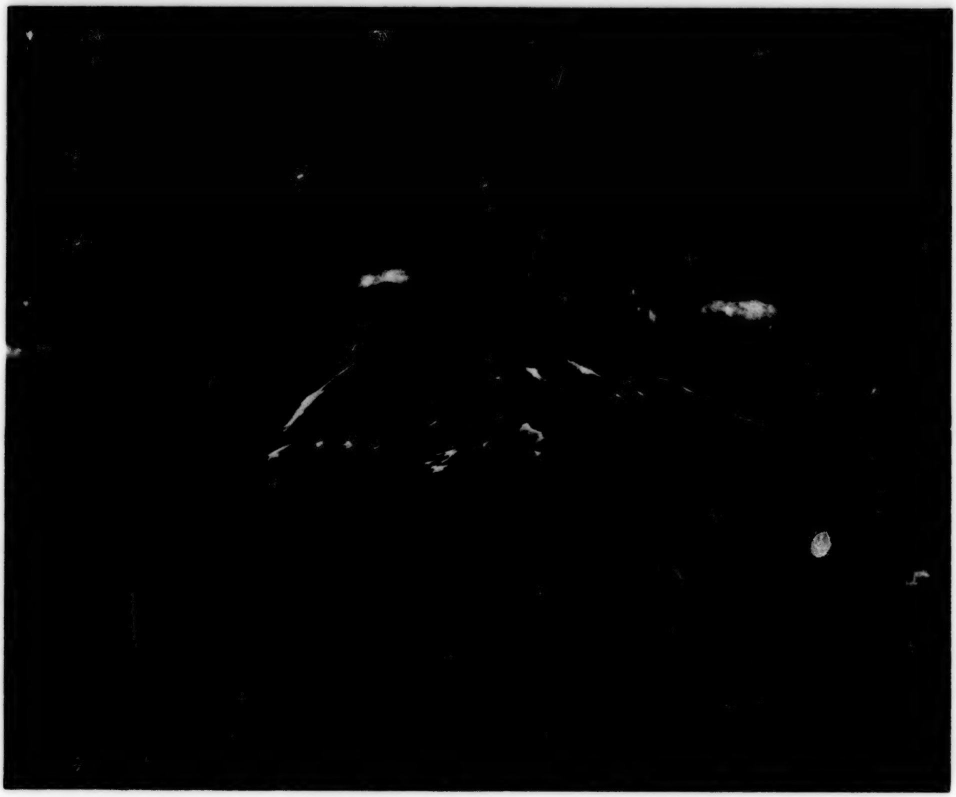
EXAMPLES OF RESOURCE INVENTORY AND MONITORING IN NATIONAL PARKS OF CALIFORNIA

Proceedings of the Third Biennial Conference



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Cover photograph: View of Lassen Peak, Lassen Volcanic National Park, California. *(National Park Service photo by Bruce Hawkins)*

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Examples of Resource Inventory and Monitoring in National Parks of California

Proceedings of the Third Biennial Conference
on Research in California's National Parks
September 13-15, 1988
University of California
Davis, California

Edited by:
Charles van Riper III
Thomas J. Stohlgren
Stephen D. Veirs, Jr.
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As the Nation's principal conservation agency, the Department of the Interior has responsibility for most of our nationally-owned public lands and natural and cultural resources. This includes fostering wise use of our land and water resources, protecting our fish and wildlife, preserving the environmental and cultural values of our national parks and historical places, and providing for the enjoyment of life through outdoor recreation. The Department assesses our energy and mineral resources and works to assure that their development is in the best interests of all our people. The Department also promotes the goals of the Take Pride in America campaign by encouraging stewardship and citizen responsibility for the public lands and promoting citizen participation in their care. The Department also has a major responsibility for American Indian reservation communities and for people who live in Island Territories under U.S. administration.

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Preface

The papers in this book were selected from the 81 presentations given at the Third Biennial Conference on Research in California's National Parks. The overall theme for this meeting was "The Integration of Research into National Park Service Resource Management Decisions." The conference was held at the University of California, Davis, on 13-15 September 1988, and was sponsored by the National Park Service Cooperative Park Studies Unit and the Institute of Ecology at the University of California, Davis.

This book highlights research and resource management efforts to inventory and monitor natural and cultural resources in the national parks of California. Many of the inventory and monitoring protocols and techniques should be applicable to other national park areas throughout the United States, and possibly throughout the world. The papers in this book naturally divide themselves into three major sections: 1. Fauna; 2. Flora; and 3. Physical Processes. Each section is prefaced with a brief introduction that integrates information in the subsequent papers of that section. The majority of the papers in this proceedings deal with the inventory and monitoring of animals and vegetation, but there is also a section on monitoring techniques for fire management and physical resources. Each paper represents original research and has been peer reviewed by at least two professional leaders in that particular research discipline.

This book is a direct result of the efforts of numerous agencies and individuals. In the planning stages, National Park Service Western Region scientists provided advice and assistance in structuring the conference format University of California, Davis graduate students, directed by M. K. Sogge, assisted with logistics at the meeting, operating slide projectors and providing general assistance with the paper sessions. We thank Jim Wood for his editorial scrutiny of the book, and the many reviewers who unselfishly devoted their time and efforts to improving each chapter.

Financial support for this publication was provided with matching funds from the Western Region Resource Management and Research program and the Washington Office Servicewide Publications Program of the National Park Service. We thank Bruce Kilgore, Dave Cherry, Gene Hester, and Donna O'Leary of the National Park Service and Peter Richerson of the UC Davis Institute of Ecology for their support throughout this book's development. We are also deeply indebted to all the talented individuals who gave their valuable time in seeing this product through the publication stage.

Finally, this book, like the products of other symposia focusing on particular problems, should help to redirect and improve the quantity and quality of research and resource management actions in an area of presently active interest within the National Park Service. If this book acts as a stimulus for additional legislative demands and commensurate funding for work on inventory and monitoring of national park resources, then we will be enormously satisfied that our organization and editorial efforts of the past year have been well spent, as have those of the authors.

September 1989

CHARLES VAN RIPER III
THOMAS J. STOHLGREN
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SILVIA C. HILLYER
Editors

Foreword

The papers in this volume are contributions from scientists, students and resource managers. The focus of their studies is on the inventory and monitoring of the natural resources of National Park Service areas in California. Support for these studies came from the authors, from individual parks, universities, and the Western Region and Washington Office programs of the National Park Service. The rich variety of the 81 presentations given at the conference and the 18 papers included in this proceedings reflects the diversity of science in the National Park Service. I applaud the efforts of the contributors. With modest funding and a broad base of public and institutional support, they have pursued important lines of work in and about the national parks and monuments in California.

There is much to be done. As a people, we face the prospect of extensive local and global environmental changes which have and will buffet the natural resources of our parks, resources which we are committed to protect from change due to modern human influences. To protect these park resources as representative examples of natural ecosystems and as baselines for measuring future change, we must increase our efforts to develop basic information, to inventory and monitor our park (and global) resources, to quantify and evaluate the changes, and to design resource management programs that will maintain and restore those resources. We must develop the information necessary to alert our managers, our leaders and our people to the importance of their natural surroundings as elements of those basic resources which sustain us, inspire us and represent our natural biological and environmental heritage.

The papers which follow illustrate just a few of the problems which we face, and some of the methods being used to monitor and evaluate them. We need much more.

November 1989

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SECTION I: FAUNA

Introduction to the Section on Fauna

We begin this volume with a series of papers that deal with inventory and monitoring of vertebrate animals in the national parks. Vertebrates liven the scenery for many park visitors and traditionally have generated considerable interest for park visitors and problems for park mangers. However, as will be seen in the following papers, many parks do not have adequate inventories of the animal groups contained within their boundaries.

The papers in this section will acquaint the reader with efforts to inventory and monitor some of the major faunal groups found in National Park System areas in California. An introductory paper by Quinn and van Riper sets the stage for this book by identifying many of the deficiencies that presently exist in the inventory and monitoring data bases from park areas in California. They also show that problems in interpretation of existing data can occur when dealing with an inadequate or incomplete inventory. Certainly not the first to be aware of this problem, these authors have provided examples of poor inventories which might be rectified.

In the second paper, Sauvajot et al. use national park avifauna species lists as an example of the inadequacy of presently existing data bases. Birds are one of the most visible animal resources in a park, and numerous visitors come each year to specifically watch them. One would expect, therefore, that information on the status of bird species in the national parks should be one of the most complete. According to Sauvajot et al., this is not the case. Even in California, which has a number of highly visited parks, they found that no two parks had the same abundance and status categories for their bird checklists. In fact, two of the 16 parks did not even have a published bird checklist. When these authors made biogeographic comparisons between parks, they found that the lack of a consistent inventory format kept them from drawing any but the most general conclusions.

The techniques and efforts needed to establish baseline inventory information vary with the group of birds under study. One group of birds that is very difficult to inventory and then monitor are the raptors. These species often nest in inaccessible locations, have a long life span, cover great distances during a single day, have large home ranges, and are difficult to see. In the third paper, Robert F. Plantrich outlines the techniques that Santa Monica National Recreation Area uses in surveying and monitoring golden eagles. He shows the advantages of helicopters over fixed-wing aircraft for monitoring eagles; helicopters have also been successfully used to monitor peregrine falcons (White and Sherrod, 1973; Monk et al., 1989).

Another segment of the avian community that is extremely difficult to inventory and monitor are rare and endangered species. Even within this group of difficult subjects, the marbled murrelet, described by Ralph et al. in the fourth paper, is one of the most difficult. They point out that only two nests of this species have ever been found in California, and its distribution in California and elsewhere was poorly known. The monitoring technique used by Ralph et al. to census this rare species is novel and can serve as a basis for other studies.

Moving to mammals in the next paper, Cook et al. found many problems similar to those described in the earlier papers on birds. They show that mammalian inventories, although not quite as bad as avian inventories, are often inadequate and have led to serious misinterpretations (Newmark, 1987; Quinn et al., 1990).

Monitoring techniques for mammals are considerably different than those utilized for birds. In the next paper, Chow et al. show how they worked with a difficult and endangered species, the "bighorn" or mountain sheep. They present some of the biological problems likely to be encountered when reintroducing extirpated species into former habitat, including national parks. The extensive use of radio telemetry to monitor animals is one of the strengths of their experimental design. This technique, although initially expensive, is often the only way many species can be properly monitored.

In the paper by Salamunovich and Ridenhour, techniques for monitoring aquatic resources are discussed. These authors deal with salmon and trout in Redwood National Park. Their objective was to determine if management actions were detrimental to the fish. The work required monitoring young salmonid populations in an estuary, following growth, feeding habits and other factors leading to an understanding of the fundamental importance of the estuary and the impact of efforts to artificially limit late summer water levels in the estuary.

The last paper in this section deals with people. The paper by Lee et al. describes a visitor perception survey of potential crowding and annoyance at Whiskeytown National Recreation Area. The experimental design of their study would be appropriate for any national park that had a potential for visitor crowding. Their findings were somewhat surprising; in only a few areas did people feel crowded. These areas were the open campground and the open beaches. The authors provide simple, cost-effective recommendations for the park to follow to alleviate these problems in the future. Planting shrubbery around the isolated campsites and posting signs at several of the beaches should, they feel, alleviate most of the problems.

In summary, the papers in this first section provide a broad spectrum of inventory and monitoring techniques for use in parks and other natural areas. It is evident that the continuing interplay between basic studies and their use in the management of national park resources is a critical element in the long-term preservation of the parks and their biological diversity and other natural values. Good inventory and monitoring programs are essential to this effort.

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Design Considerations for National Park Inventory Databases

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INTRODUCTION

One critical function of national parks is the long-term maintenance of biological diversity. Accurate assessments of the occurrence of species or genotypes (or perhaps more importantly, their absence) in parks, and the health of individual protected populations, are essential to both the management of diversity within existing parks and the identification of new sites for acquisition that encompass biotic diversity that is not now effectively protected. As the oldest and most extensive park system in the world, U.S. national parks provide the principal model for conservation efforts in countries where the erosion of biotic diversity is even more pressing. Planning success within the U.S. National Park Service (NPS) is, therefore, likely to have ripple effects on global biodiversity conservation efforts.

Unfortunately, the state of biological inventories and monitoring in U.S. national parks is distressingly incomplete and fragmented. Perhaps the most glaring example of the inadequacies of readily available data is provided by the widely cited study of Newmark (1987). Newmark surveyed data, including NPS species lists and publications, and interviewed park personnel on the status of larger mammals (specifically lagomorphs, artiodactyls, and carnivores) in 12 western national parks. A number of these mammalian species are known to have disappeared because of past hunting or predator control. But he inferred that 39 populations, more than three species per park, had disappeared from "natural causes" since the parks had been established. Newmark and others have concluded, therefore, that existing national parks were incapable of protecting even the most conspicuous and popular species that they were meant to conserve. Fortunately, most of these "extinctions" appear to be due to poor inventories, or at least lack of accessible sighting records, rather than actual disappearances. On reexamination of the available information, Quinn et al. (1990) concluded that no more than three of the 39 apparent "extinctions" represented actual disappearances of the animals from the parks and their immediate environs. Obviously, the continued existence of the putatively extinct populations is good news. On the other hand, the fact that the controversy could arise at all, especially in species as "popular" as large mammals, suggests that all is not well with present inventory information in the national parks.

In fact, the history of NPS administrative decisions and subsequent projects to inventory natural diversity is one of ambitious plans that are rarely completed. Reasons vary, but inadequate to nonexistent funding, lack of qualified personnel, and a general lack of priority treatment and systemwide support all contribute. Part of the solution will have to include greatly increased funding levels for inventory and monitoring activities. However, additional efforts are likely to be largely ineffective unless plans are well designed and coordinated, both within and among parks and regions. At present they are not. The report by Sauvajot et al. (Paper #2 this volume) provides a striking example. In the 19 California national parks surveyed for avian inventory information, no two used the same criteria for recording standard status information, such as permanence (e.g., resident, migrant), seasonal occurrence (winter, summer, year), reproductive status and abundance. Some record accidentals and vagrants, while others do not. None have numerical abundance data, much less year-to-year population changes (apart from Audubon Christmas counts). There is no standard census protocol. Therefore, abundance data cannot be directly compared from park to park. In many cases, Sauvajot et al. were unable to trace the original sources of the data. Conversations with NPS scientists familiar with inventory and monitoring studies suggest that this experience is not atypical.

Clearly, both standardization and greater resources dedicated to inventory studies are needed if inventory data are to be adequate for systemwide planning or comparative scientific studies. Standardization should also lower costs, as individual scientists would have less need to "reinvent the wheel." However, the appropriate standards are not obvious. In our experience, many scientists responsible for inventory and monitoring feel that they have received little guidance on what data are needed, or how the data are best made useful to managers, Congress and other users.

We suggest that an immediate investment in workshops or other forums to design and standardize inventory and monitoring studies will pay for itself many times over in increased effectiveness and reduced staff time. It is important to involve managers, land use planners and academicians, as well as park scientists, to identify user needs. A minimal set to be addressed includes:

1. Data required to estimate impacts of visitation and proposed changes within the parks;
2. Identifying long-term, externally driven environmental change, including effects of climatic change, water and air quality, and land use outside the parks (e.g., external effects on migratory birds);
3. Assessing the status of rare, threatened, or endangered species;
4. Strengthening interpretative programs;
5. Guiding future land acquisitions;
6. Coordinating with management of contiguous lands. (To be effective, this obviously requires at least partial compatibility with the inventory and monitoring activities of other federal and state land use and conservation agencies); and
7. Future compatibility with larger-scale regional and global environmental monitoring efforts, including EOS, proposed global biodiversity surveys, standardized federal Geographic Information Systems (GIS) (if any), U.S. Geological Survey (USGS) and National Oceanic and Atmospheric Administration (NOAA) mapping operations, etc.

In order to make inventory projects manageable, it is important that the design be relatively straightforward and targeted toward the main kinds of information needed by decision-makers and researchers. A fundamental consideration of any inventory should be how to integrate data into a GIS. The design plans must also consider the present form of existing inventory information in the park.

As with other forms of organizational information systems (Quinn, 1984), it is probably most effective to make decisions on inventory policies and procedures in an incremental fashion—beginning at the most basic level, and building on it systematically as experience and feedback from the field dictate. Effective feedback from researchers and managers requires that a working system be in place, and sufficiently straightforward in its use so that initial acceptance is not a major problem. In other words, the military prescription "KISS" ("Keep it simple, stupid") can be profitably applied to the initial stages of major inventory projects.

The most basic decision in inventory database design is to determine what information should be included: (1) initially and (2) eventually. Initial information ought to be that which is readily available and of immediate use. Obvious categories include overall species lists, current abundance, the status of particular species of interest (e.g., endangered, threatened, listed, under study), and basic habitat information (e.g., elevation, slope aspect, rainfall, gross vegetation type). A variety of more detailed information types might eventually be added. Examples include censuses from permanent transects, fates of marked individuals, and detailed records of individual sightings or collections. Presumably, the long-term design will have to be somewhat hierarchical, with the basic information from each park tied into more detailed records in those cases where they are available. Automatic updating of the database from new field records and extensive cross-validation (e.g., to check for data entry errors and to keep taxonomic use consistent) are also necessary to maintain the integrity of any extensive inventory system.

METHODS

Since 1985, we have begun constructing a simple database that meets some of these needs and design constraints. We have recorded species occurrences of birds, mammals and vascular plants in national parks, state parks and other discrete conservation areas in California (with a few current entries from other western states). The database is currently stored at the University of California, Davis DES/CPSU computer facility and has entries for 19 federal park units, and roughly 70 state and private parks and reserves in California. Species lists were obtained by directly contacting responsible park officials. Where available, alternate species lists were also obtained from other sources, including the NPFLORA database for plants in national parks, the California Wildlife Habitat Relationships (WHR) data base, the compilation of mammals in national parks assembled by Newmark (1986 and unpublished) and Audubon Christmas counts. We also directly interviewed knowledgeable field workers in the parks and in universities. All data are recorded in dBase III+ format files, tied together using dBase's relational database capabilities.

While this program only meets the most simple of the objectives outlined above, the data represent a basic level of information that ought to be available for all parks. We have standardized their format and taxonomic usage. They include inventory data from half a dozen federal and state agencies, among whom there has been little previous formal cooperation in inventory studies. By concentrating on a single region, a usable database can be sufficiently complete to be useful with a manageable investment of funds and researcher time. In its present form, we have used it for testing published hypotheses in ecology and conservation biology. It is available as a source of standardized species lists for cross-validation by agency researchers and as inputs into regional GIS programs. We hope it will be a catalyst and source of outside support for more sophisticated studies in individual parks, which in turn, if successful, may help develop a cooperative interagency regional biodiversity database useful as a model for national and international efforts.

RESULTS

The current database consists of species lists for each park, master lists of bird, mammal, and vascular plant species, lists of synonymies that have arisen to date, and a list of parks and park characteristics (e.g., location, area, altitudinal range). We are in the process of adding physical environment and gross vegetation-type data to each park list. For most parks we have data for birds, mammals and plants. A few parks reported data for other groups, including reptiles, amphibians, freshwater fishes, marine invertebrates and algae.

Both the kinds and the quality of the data that we received were quite variable and are described elsewhere in this volume in the papers by Cook et al., Robinson et al. and Sauvajot et al. In most cases, only presence-absence data are recorded. If available at all, relative abundance data are inconsistent from park to park—see discussion by Sauvajot et al. in this volume. Although these data are in the database, considerable standardization among parks is needed if they are to be useful for comparative analyses.

Within each park, species of birds, mammals and plants are recorded at the levels of order, family, genus, species, subspecies or variety, and common name (if any). These are tied into a master list consisting of all species reported in the study to date. The master lists are maintained using the nomenclature of standard taxonomic references (American Ornithologists' Union, 1983 for birds; Hall, 1981 and Jones et al., 1986 for mammals; and Munz and Keck, 1968 for plants.) Other species characteristics (e.g., range, legal status—see above) can easily be attached to the master list, and thus to the much more extensive park lists. The master list can also be used to create an electronic checklist for data entry, which serves both to make the process much faster and to prevent typing errors.

Also related to both the park and master lists are taxonomic synonym lists. Simple utilities using these lists identify and convert taxonomic synonyms to the standard current usage, and flag unknown names for further investigation. These procedures also allow for almost automatic future taxonomic updates as new standards are published.

Finally, the park list contains names and standard abbreviations for parks, their location, area, date of establishment and a variety of habitat information. Important and obtainable habitat variables include elevations, vegetation types, aspects, rainfall, etc. As GIS databases become more common in parks, each inventory list can be connected to the GIS. The standardized data format and software should make future upgrade paths straightforward.

DISCUSSION

This study may be viewed as a pilot project and has addressed only a subset of the kinds of data that are needed to establish an effective regional inventory system. Even so, we have had to make repeated procedural decisions that are central to any biodiversity database, reflecting the lack of standardization in nomenclature, sampling protocols, and record-keeping conventions. These usages desperately need to be decided and standardized among researchers and users, and preferably across agencies. This can only be done through extensive consultation among scientists and users, and ought to be on the agenda for much-needed interagency workshops on inventory and monitoring strategies. Particularly important considerations include:

1. What are the appropriate taxonomic references and validation procedures?
2. What specific provisions should be made for officially listed (endangered) species? (In particular, provisions should be made for compatibility with data collections emphasizing endangered species, such as the National Heritage database.)
3. Should there be any special treatment of species which are locally rare or endangered (i.e., marginal populations), but which are common elsewhere?
4. Is park-wide habitat characterization (e.g., altitudinal range, weather, vegetation types) adequate, or should species occurrences be recorded on a habitat-by-habitat basis?
5. Should ecological communities be identified? (In contrast to a number of other efforts, such as NPFLORA, we believe that the initial answer to this question is no. There is little consensus among ecologists on how to best identify distinct communities, or even of whether the community concept is particularly valid. We suggest that it is initially more useful to concentrate on specific species occurrences, which are less dependent on subjective interpretations. Community types, if needed, can likely be later derived from species-abundance data or the occurrences of indicator species.)

If inventory data are to be useful for regional or national research and decision-making, it is imperative that data be collected and kept in as consistent a format as possible. The design should be able to access poor inventories for comparative purposes, yet be able to take better data into account when available. A basic regional species inventory ought to have at least the following features:

1. A master species list for each taxon (e.g., birds, mammals, vascular plants) and synonym lists. Master lists can be used to tie the database to standard taxonomic references and reference collections, and to record species-specific data (e.g., range, diet, legal status). The existence of an authoritative reference list will greatly facilitate data entry and error checking. Synonym lists facilitate consistent taxonomic uses, provide automatic updates, and could also be used to query about common identification errors (e.g., "Caution—the very similar Dusky Flycatcher is more probable in your region.")
2. Consistent criteria for listing species. For example, decisions must be made on how to treat vagrants and accidental occurrences, permanent versus part-time residents, and feral or horticultural species.
3. Consistent reporting categories. At present there are no established standards for reporting species densities, residence status (e.g., fall-spring migrants), reproductive status (e.g., does the species reproduce in the park), or population dynamics (e.g., slowly increasing since 1960). Reasonable standards should be adopted and incorporated into standardized sighting cards. We suggest that minimal reporting standards should include:
    a. estimates of relative abundance on a standardized scale;
    b. seasonal status for part-time residents;
    c. breeding status;
    d. indication of the observer(s) and probable validity of the observations;
    e. special circumstances or comments; and
    f. location of any specimens or photographs.
4. Cross-referencing to a bibliographic database, including technical literature reports and student theses done in parks, as well as research publications from the parks and general ecological and systematic literature.

The usefulness of any database depends upon its ease of use. It is, therefore, worthwhile to conduct any such project using widely used hardware and software, minimizing both equipment and training costs among users. It is equally important that products of the project be regularly updated and distributed to users. Finally, there should be a reasonable upgrade path as the database becomes more extensive, and better hardware and software become available to dedicate to database uses. With these considerations in mind, our project has used standard IBM-compatible personal computers and dBase III format data files. Text file formats (e.g., WordPerfect) and communications protocols (e.g., Sea Dog, Common) can also easily be standardized to conform to agency usage.

The species list database structure described above and initial entries from California are currently virtually completed. Although species occurrences from the most recent studies have not been incorporated in all cases, the underlying structure makes the updates completely straightforward. We anticipate that the updating of the central database will be an ongoing process, but will require relatively little effort (e.g., it can easily be handled by a part-time employee).

The next step in the process is to add a level capable of detailing actual sightings or collections related to the species lists in parks, and to provide for automatic updating from individual occurrence records. This is straightforward from the point of view of programming, but will require greater standardization of sighting or collection records in parks to achieve maximum usefulness (see recommendations below). Finally, the connection to GIS capabilities needs to be established.

As the degree of detail increases, greater attention to sampling design will become increasingly essential. We view the optimum approach as consisting of a hierarchical strategy in which all parks (and eventually other conservation lands) will be expected to establish a base level of inventory information, which might include species and species status lists, GIS standardized vegetation type maps discernible from aerial and satellite imagery, and basic physical environment information maps, such as rainfall, temperature, altitude, and perhaps air and water chemistry. Parks and other units with adequate resources might add, in a preplanned stepwise order, increasing levels of detail on one or more of the basic information categories. This hierarchical approach would not only provide procedural guidance and tools for individual scientists, but would also generate a nested database in the spirit of a stratified experimental design, in which the better studied parks serve to verify and help estimate confidence intervals on the more basic data from the less well inventoried parks and other conservation lands.

CONCLUSIONS

We suggest the following approach for strengthening the inventory and monitoring efforts of the parks in the region:

1. There should be a meeting of park and university researchers to design a specific common format for reporting inventories in California. Although it is important that the group be of a manageable size, we believe several participants from other agencies could profitably be involved. It is clear that a well-thought-out regional plan is likely to be adopted as a model by other agencies and regions.

2. A product and software should be published and made available to all parks in California and, as appropriate systemwide, on a common inventory format for NPS resources. Software should be designed to simplify entry and prevent errors compatible with systems currently in parks,

including GIS systems. Compatibility with databases maintained by other state, federal, and private agencies should be explored.

3. A common reporting format for researchers and contractors should be established for future projects, theses, and contracts. In particular, reporting in common format should be required as a condition of approving and/or funding future research.

4. There should be a central location for all inventory databases. A university/CPSU is a logical choice because of its regional service function and the availability of research associates. However, capabilities for remote access and updates are needed, and the center needs long-term guaranteed support.

5. A clear organizational responsibility for organization, management, and maintenance of the database should be established.

6. Mechanisms for feedback from parks, planners, and researchers must be built into the system.

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Comparative Analyses of Bird Inventory Databases from California National Parks

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Abstract. Assessing the effectiveness of biological diversity preservation within nature reserves necessitates the availability of accurate and comparable inventory databases. In compiling bird inventory data for 16 California national parks, we found that often these data are not presently comparable. Bird inventories show considerable variation in species numbers between parks, depending on types of species considered (total species, regularly occurring species, resident and nesting migratory species, nesting species, and resident nesting species), presumably reflecting not only biological differences, but also differences in data availability and reporting formats. Only half of the 16 parks report information for all species types, and no two parks report data in the same way. With so few parks having adequate baseline avian inventories, comparative analyses are extremely difficult. We suggest that a standardization of methods is needed to improve inventory databases, and stress the importance of interagency cooperation in making such standardized data available so that critical conservation questions can be addressed properly.

INTRODUCTION

As human development and habitat modification accelerate, increasing interest has been focused on developing strategies to preserve biological diversity (Wilson, 1988; Tangley, 1988; Greene and Losos, 1988; Roberts, 1988; Office of Technology Assessment, 1986; Soulé, 1986). One necessary component of such programs is the availability of accurate and comparable inventory databases of species diversity. Such information is essential to help resolve current conservation issues. For example, the "SLOSS" debate over the best allocations between single large versus several small reserves (Simberloff and Abele, 1982; Gilpin and Diamond, 1980) and the controversy over historical extinctions in national parks (Newmark, 1987; Quinn et al., in press) depend upon accurate inventories.

We have worked to develop the data necessary to effectively address these types of issues by carefully collecting, standardizing, and reviewing avian inventory information from California parks and reserves. In particular, we obtained bird species lists from a number of California park and National Park Service units in an attempt to both compile the best inventory information available from these units and to develop a readily accessible, comparable and useful database for answering conservation questions pertaining to avian biodiversity.

This paper presents the results of a preliminary analysis of comparability between the 16 National Park Service units in California (parks, monuments, recreation areas and seashores, all collectively referred to as parks) for bird species inventory data that we obtained. National historic sites have not been included in the analysis. We stress that these results are only preliminary, reflecting bird species information that we currently have on record, and recognize that our database may not necessarily contain all of the current information available from each park. By presenting this initial report, we hope to demonstrate some of the difficulties we encountered in our work and to initiate feedback from individual parks, organizations, and agencies to improve these data and to standardize database designs. A further discussion of objectives and database design is found in the paper by Quinn and van Riper (this volume).

METHODS

Bird species inventories were obtained by contacting park officials from 16 national parks in California. All parks were contacted in 1985 and many have provided updates since that time. Table 1 contains the parks, park codes, and references for each list, including the years of most recent known revisions. Most species lists are checklists, such as those available to park visitors. However, some parks provided other types of resource inventory data or additional information on avian species within the parks.

TABLE 1. The 16 California National Park Service units that provided lists used in this study. "List Reference" includes names of authors or individuals compiling list along with year of list publication or most recent known update.

Park Name	Code	List Reference
Cabrillo NM	cabr	Edwards 1979*
Channel Islands NP	chis	Jones et al. 1985
Devils Postpile NM	depo	Madeiros 1982, 1987
Death Valley NM	deva	Norris and Schreier 1982
Golden Gate NRA	goga	White 1974*
Joshua Tree NM	jotr	Freilich 1982
Lava Beds NM	labe	Escano 1972
Lassen Volcanic NP	lavo	Hatch 1982*
Muir Woods NM	muwo	(author and year unknown)
Pinnacles NM	pinn	Avery and van Riper 1987
Point Reyes NS	pore	Shurford 1982
Redwoods NP	redw	Lester 1977*
Santa Monica Mountains NRA	samo	1985 park data*
Sequoia and Kings Canyon NP	seki	Norris 1984
Whiskeytown NRA	whis	Wintu Audubon Soc. 1981*
Yosemite NP	yose	Gaines et al. 1983

*Unpublished data

Each species list was placed into a standardized format, and nomenclature modified (when necessary) to agree with the 1983 American Ornithologists' Union's *Checklist of North American Birds* (1983). Data were then entered into dBASE III computer files, so that all park inventories could be easily accessed and readily compared.

In order to conduct species diversity comparisons effectively between the parks, it was necessary to develop consistent analysis categories reflecting the varying frequencies of occurrence for each avian species within the parks. For example, while one park may provide a very detailed list of every species, including accidental and irregular migrants, another may provide only information on common resident species. To compare among parks, some form of "common denominator" or analysis category had to be utilized. In addition, specific conservation issues which focus on particular types of inventory data (e.g., only breeding species) also require such categorization.

For the purposes of this preliminary report, we defined five different analysis categories for inter-park diversity comparisons, including: (1) total species; (2) regularly occurring species; (3) full-time resident and nesting migratory species (where migratory species may include both part-time residents and "true" migratory birds); (4) nesting species; and (5) full-time resident nesting species (not all full-time residents were listed as nesting).

Although these definitions are somewhat arbitrary, we feel that they reflect the type of information important for assessing bird species conservation issues and provide increasing levels of confidence as to which species consistently use specific national park areas. Further consultation is needed to standardize these categories, or analogous usages, among researchers in individual parks.

We categorized each bird species from each park using abundance information, residency data, and nesting records provided by the parks (see, for example, Table 2 showing abundance information). It is important to note that, although we defined the five analysis categories, inclusion of specific species from each park into these categories was deter-

TABLE 2. Abundance classifications used by each park for bird species. Note that no two parks use identical codes and all provide varying amounts of data.

Park	Abundance Information
cabr	*
chis	a c u r x
depo	(no additional data)
deva	C U R Ca A
goga	(no aditional data)
jotr	C FC U R CAS IRR
labe	r irr ?
lavo	c u r x ()
muwo	A CM FC U R
pinn	A C U R
pore	A Cm Fc U R Cs X
redw	C R Cas Ac?
samo	C
seki	C FC U R Ca E
whis	c fc u r cs a *
yose	c-a u-fc r vr irr

cabr: * = rare, accidental, or inconclusive; chis: a = abundant, c = common; u = uncommon, o = occasional, r = rare, x = accident; deva: C = common, U = uncommon, R = rare, Ca = casual, A = accidental; jotr: C = common, FC = fairly common, U = uncommon, R = rare, CAS = casual, IRR = irregular; labe: r = rare, irr = irregular visitor, ? = hypothetical; lavo: c = common, u = uncommon, r = rare, x = extremely rare or accidental, o = hypothetical; muwo: A = abundant, CM = common, FC = fairly common, U = uncommon, R = rare; pinn: A = abundant, C = common, U = uncommon, R = rare; pore: A = abundant, Cm = common, Fc = fairly common, U = uncommon, R = rare, Cs = casual, X = accidental; redw: C = common, R = rare, Cas = casual, Ac = accidental, ? = no verified observation; samo: C = casual; seki: C = common, FC = fairly common, U = uncommon, R = rare, Ca = casual, E = extraordinary; whis: c = common, fc = fairly common, u = uncommon, r = rare, cs = casual, a = accidental, * = assumed to occur but no confirmed sightings; yose c-a = common to abundant, u-fc = common to fairly common, r = rare, vr = very rare, irr = irregular.

mined entirely by data provided by the parks. Appendix A includes all species data used in our analysis.

After categorization was completed, we examined bird diversity patterns between the parks, making preliminary assessments of park data comparability for each of five analysis categories.

In some parks, more complete data and updates apparently exist. We have, therefore, designed the database and analysis so that additional data can be incorporated with minimal effort, and analyses rerun at any time. Until the database is completely current, the analyses are obviously preliminary, but still serve to show many of the patterns and potential problems that parks might expect to experience in the future.

RESULTS

Total Species

In the first analysis step, we examined total number of species. For this category, a species was included if it was indicated on a park list. Figure 1 illustrates the results of this step, showing total number of species, with parks arranged from smallest to largest on the horizontal axis. At this level of data resolution, a fairly optimistic interpretation of species diversity as related to potential conservation is possible. Fully 86% of all California bird species are found in the 16-park assemblage (based on Binford's 1986 California bird checklist), with 483 of the total 562 recognized California bird species recorded within all combined parks. In addition, among the 14% (79 species) which do not occur within any park, 90% of these (71 species) could be considered "accidental" for the entire state (see Binford, 1986). However, is it justified to make such sweeping assessments based on these "total species" data?

It is clear that enormous variability exists between park data bases in both species counts and the types of species included. For example, Point Reyes contains 412 species, including a large number of very uncommon migrants, while Devils Postpile lists only 64, most of which are considered common species. Muir Woods, adjacent to Point Reyes (but much smaller and with much less habitat diversity) contains only 58 species. Cabrillo, on the other hand, is the smallest park in the assemblage and lists 269 species.

Thus, inter-park comparisons may not be appropriate at this level, especially if parks use different criteria for listing birds. Considering the wide variability in species number, this appears likely, and is further supported by examining the number of "irregular bird species" or those that are unlikely to consistently occur within the park on at least an annual basis (Fig. 1). Many parks show a large number of these irregular species. For example, Point Reyes lists 149 irregular species (36% of its total diversity), Death Valley 151 species (42% of its total diversity), and Cabrillo has

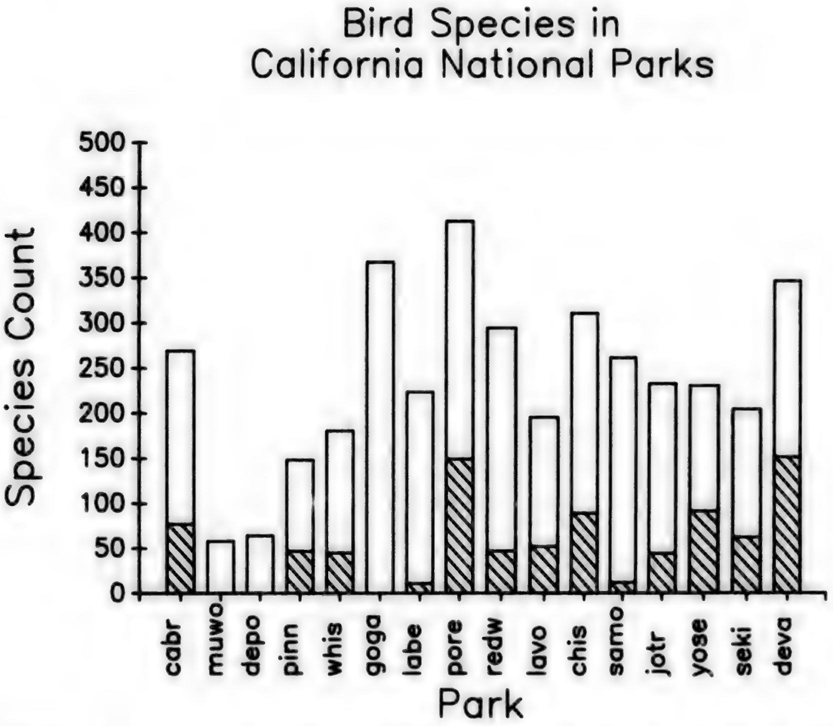

FIGURE 1. Total species diversity (total number of species) for birds in each park. Parks are indicated by 4-letter codes and arranged from smallest to largest along the horizontal axis. "Irregular bird species" are plotted as hatched subsets of total species diversity in each park.

77 irregular species (29% of its total diversity). On the other hand, some parks indicate very few or no irregular birds on their lists (e.g., Muir Woods, Lava Beds). Therefore, to make the data more comparable, and to obtain a greater degree of confidence as to which species consistently use the parks, irregular species were removed in our analysis of regularly occurring species.

Regularly Occurring Species

In the analysis of regularly occurring species (Fig. 2), two important points emerge regarding park comparability and conservation assessment. First, it is clear that removing irregular species can influence interpretations of avian diversity in the park system. The number of species was reduced by an average of 60 birds for each park; however, some parks lost no species, while others lost many (e.g., Death Valley, 151 species lost; Point Reyes, 149 species lost). As a result, if regularly occurring species are used instead of total diversity data (analysis category 1) to answer the same types of questions, it is possible that very different answers would result. For example, species-area relationships examined using regular species indicate essentially the same species diversity in both the largest and smallest parks (Death Valley with 195 and Cabrillo with 192), a surprising result that may not be true for other levels of analysis.

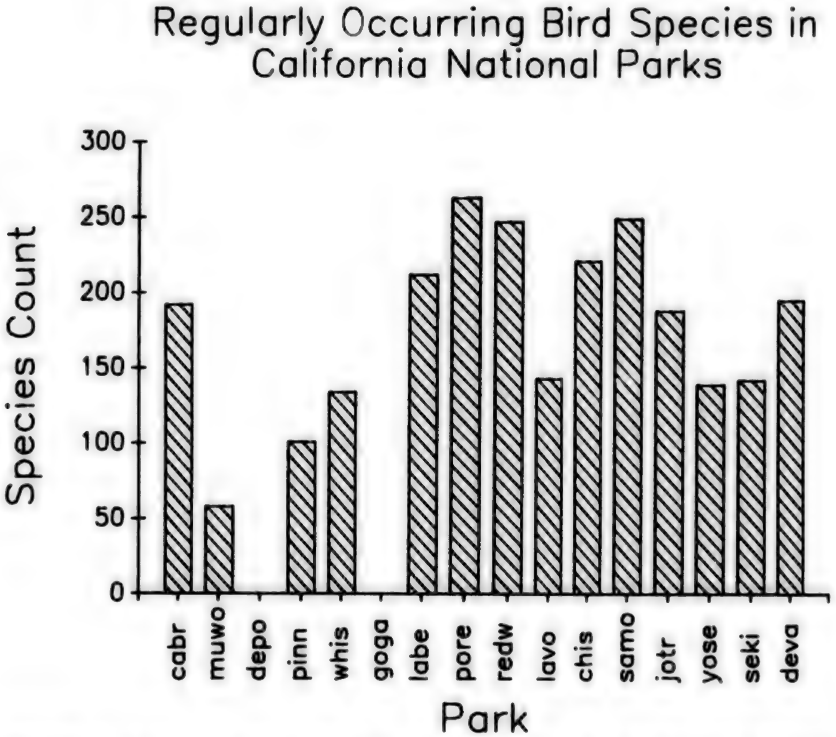

FIGURE 2. Regularly occurring species in each park. Note apparent "absence" of species from Devils Postpile (depo) and Golden Gate (goga), reducing the total sample size to 14.

The second important consequence of this refinement in data resolution is a reduction in the total number of comparable parks. Golden Gate and Devils Postpile had no regularly occurring species (Fig. 2), because neither park list provided data suitable to differentiate between regular and irregular birds. Thus, these parks could not be included in this analysis category, reducing the number of comparable parks to 14.

Lists of regularly occurring species can still contain a potentially large number of migratory or vagrant species, using the parks as "stepping stones," in addition to a number of non-nesting part-time resident birds.

While these species are clearly an important component of a park's overall diversity, it is useful to look at species which more directly depend on park resources for full-time residence and/or nesting. We, therefore, defined our third category to include only year-round residents and nesting migratory species.

Resident and Nesting Migratory Species

The resulting species diversity plot (Fig. 3) at this level demonstrates even more difficulties with data comparison. Again, the sample size of comparable parks is reduced. Not only did this analysis delete Cabrillo (with no data for bird residency), but five other parks were also eliminated due to lack of information on nesting species (indicated in Fig. 3 by asterisks). Comparative work becomes increasingly limited with only eight parks providing all the data necessary for a complete avifaunal assessment.

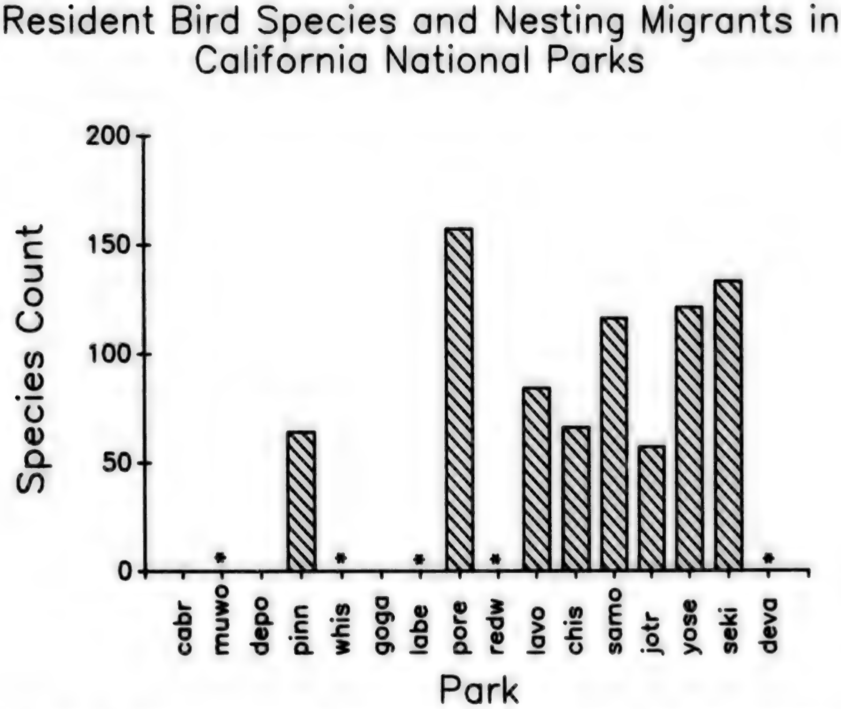

FIGURE 3. Full-time resident and nesting migratory bird species in each park. Migratory species include both part-time residents and "true" migratory birds. Asterisks indicate parks eliminated due to lack of information on nesting species.

In addition, another large drop in species diversity per park occurs as we move to this category. On average, 89 fewer species per park occur at this level when compared to the previous analysis and the range of reductions between parks is quite variable (46 to 157 species). Although such a drop is expected, it is important to note that these changes, occurring at the individual park level and within the assemblage as a whole, will ultimately influence interpretations of bird species trends derived from these data. This is especially true when declines in species diversity are a function of both ecological factors and lack of sufficient data.

Nesting Species

We analyzed current nesting species diversity, clearly an important category for conservation purposes, as our fourth step (Fig. 4). The sample size of parks providing necessary information was reduced to only nine, nearly halving the size of the original 16 park assemblage, thus limiting attempts at meaningful comparative analyses. In addition, data limitations between parks were apparent, with several full-time resident species not recorded as nesting within several parks.

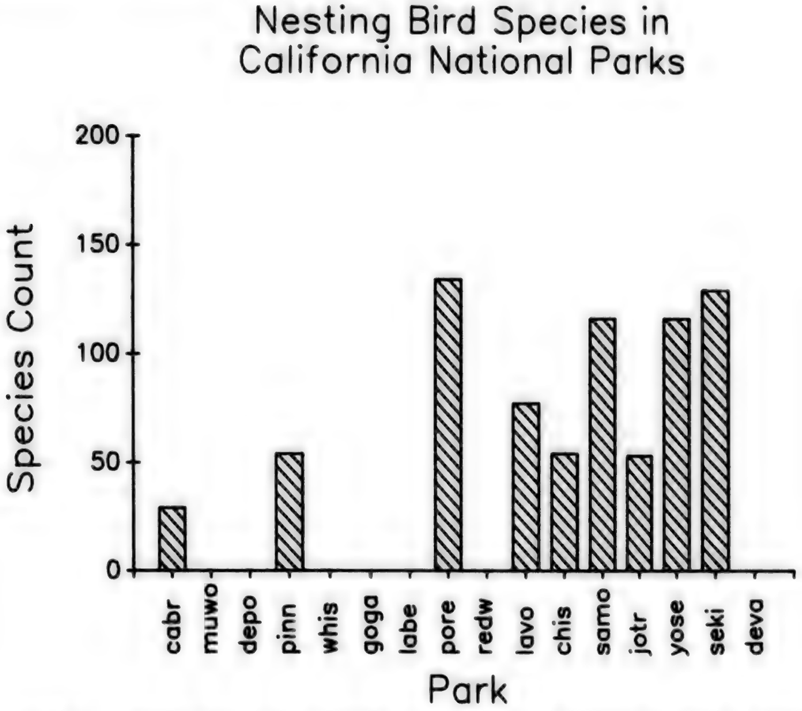

FIGURE 4. Currently nesting bird species in each park. Note dramatic reduction in sample size.

Resident Nesting Species

In our final examination of inventory comparability, we sorted the data to include only full-time resident nesting species (Fig. 5). Such birds can be considered completely dependent on park resources, and thus used to examine conservation issues, including concerns over habitat quality or island biogeographic questions. Our results illustrate once again the problems of attempting such comparative work with the existing data. Only half of the 16 parks provided data necessary for this analysis, and the important influence of categorical choice on diversity patterns will again affect research conclusions.

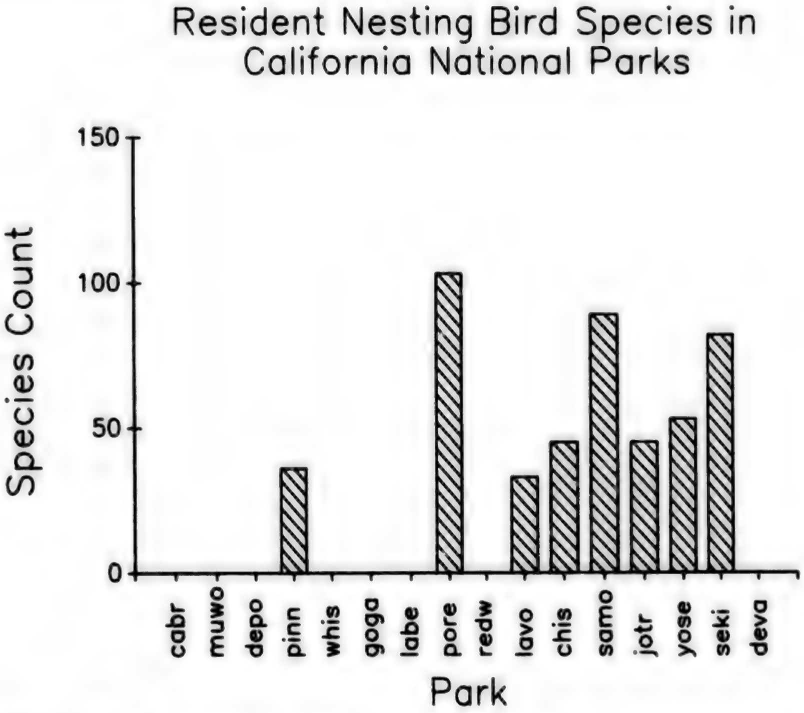

FIGURE 5. Full-time resident nesting bird species in each park.

DISCUSSION

The results of our preliminary analysis illustrate the potential complexities and difficulties of attempting inter-park comparisons using currently available National Park Service bird species inventory data.

First, it is clear that for these data, conclusions regarding bird diversity, both within and between parks, will depend on the analysis categories used. Species diversity estimates varied widely according to the type of analysis used (e.g., all species, nesting species, etc.), strongly influencing overall diversity patterns (Fig. 6), and ultimately affecting conservation interpretations and conclusions. For example, if island biogeographic questions are approached using total species in one case, and resident nesting species in another, very different answers to the same questions could result.

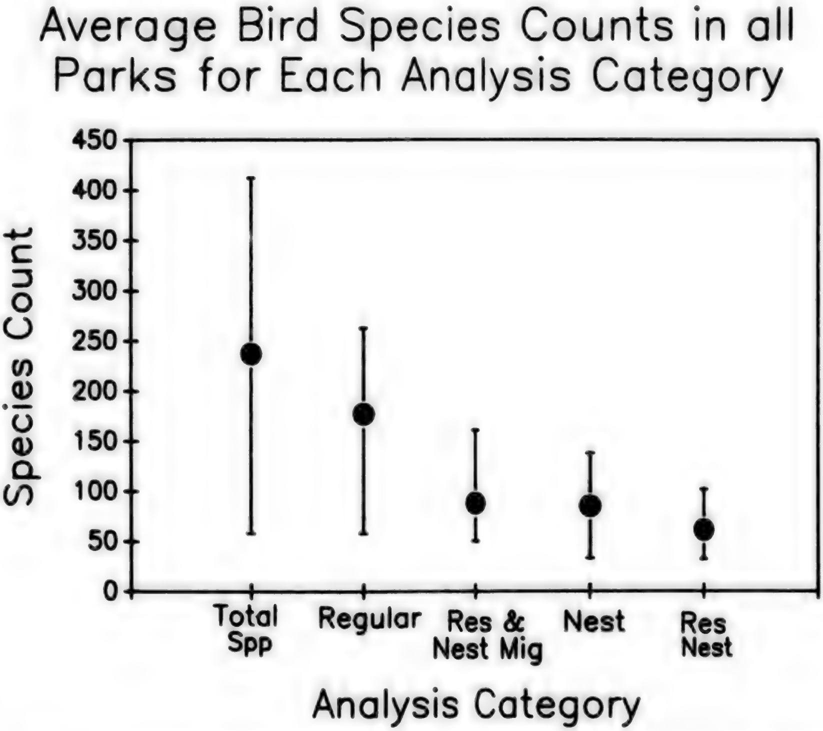

FIGURE 6. Average species counts per park for each analysis category. Error bars indicate maximum and minimum ranges of species counts from parks at each analysis category. Note the important influence of analysis category choice on overall diversity as well as the relatively broad range of species counts found among parks at each level of analysis. All calculations are based only on parks which provided necessary data.

A more important result of our analysis, however, is that the level of data resolution (analysis category) can dramatically reduce the number of parks suitable for analysis (Fig. 7). Most parks do not provide information at all levels of analysis, and thus must be excluded from some analysis categories. Comparative work becomes increasingly difficult as sample size is reduced, thereby limiting the capacity for generalized conclusions. For example, reasonable conjectures about nesting bird species conservation in California national parks is impossible when only nine out of 16 parks provide necessary data.

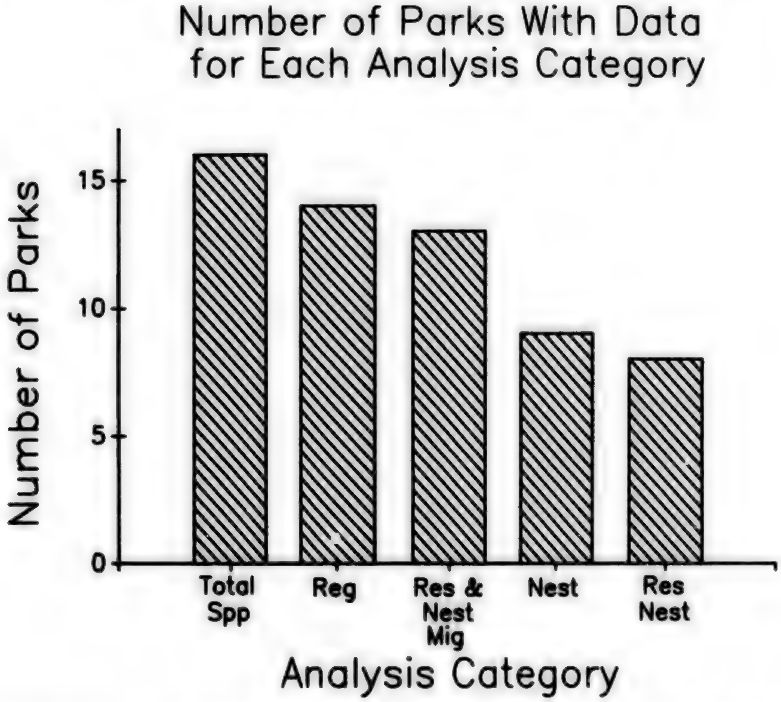

FIGURE 7. Bar graph illustrating total comparable park sample size for each analysis category.

Another difficulty, not yet considered but of clear importance, is the placement of species into specific analysis categories. Such categorization is essential for comparative analyses of bird species inventories, because many species exhibit high mobility and variable ranges over time. As a result, assessing presence or absence depends on considerations of both spatial and temporal scales. However, accurately assigning species to categories can be complicated. We discovered that the abundance, residency, and nesting data needed to separate species was extremely variable between parks, and no two parks reported data in the same way. Table 2 illustrates this variability in abundance information, and similarly diverse data were found for residency and nesting information as well. Clearly, developing objective analysis categories from this type of information is difficult, but it must be done if data are to be of maximal use for research or as the basis of management decisions.

An additional point concerning data availability should be noted. Historical extinction information, crucial to any effective analysis of biological diversity, is rarely available. In fact, only three of the 16 parks give any indication of possible historical extinctions (Joshua Tree, Point Reyes, and Santa Monica Mountains). Information on the extent and causes of known extinctions in all parks is needed if sound conservation strategies are to be formulated.

CONCLUSIONS

Analysis of the comparability of bird species inventories for 16 California National Park Service units demonstrates that use of these data for assessing patterns of avian diversity and for developing specific conservation and preservation strategies would be difficult at best. Avian diversity varies dramatically from park to park depending on the type of species considered (i.e., the analysis category). This variability reflects not only presumed biological differences, but also differences in research history and data availability between parks. In addition, the number of parks with data suitable for analysis is also heavily influenced by these criteria.

To remedy this situation, a standardization of park avian inventory data is desperately needed. Species information needs to be available in a consistent format among national parks, and for many parks, the scope of inventory data must be increased. For example, a basic standardized format for bird inventory data could parallel the existing AOU checklist system (American Ornithologists' Union, 1983). All parks would minimally need to compile species presence and absence information for birds within park boundaries, and indicate whether or not particular species could be considered accidental (represented by very few records and of unlikely occurrence in the park) or casual (represented by very few or irregular records but of not improbable occurrence in the park). Additional important information would include residency status, categorizing species as permanent full-time residents, seasonal residents, or transients only occupying the park during movements elsewhere. Finally, full-time residents and seasonal residents could be further categorized to include breeding information. Because accurate and comparable inventory information is so crucial to an effective conservation strategy, immediate efforts at improving and standardizing such data should be top priorities for National Park Service funding.

Many of our results illustrate the potential difficulties and problems faced in attempting comparative analyses of bird species inventories. But it is not our primary goal to criticize the current state of affairs. We are actively involved at this time in compiling and improving National Park Service avian inventory databases, including continually updating the data to reflect additional information available from parks. Of course, these tasks cannot be successfully accomplished without close working relationships with individual parks, agencies, and other organizations. We believe a wealth of uncompiled information currently exists, and that these data can improve our present understanding of avian distributions in California's parks.

In conclusion, we see an urgent need for standardized and comparable avian inventory data. At the same time there is a parallel need for cooperation to develop such a database. University-based inventory projects, such as ours, can provide long-term support and centralized access to standardized biological databases for researchers and managers in the field. As such, we hope to supplement under-funded agency efforts, especially those difficult to coordinate because of artificially constructed regional boundaries. In turn, our effort depends upon feedback needed from biologists and decision-makers in parks on both the basic data and our analyses. The results will be a better coordination and cooperation between national park areas, universities, and other federal and state agencies, and will contribute substantially to biological conservation efforts in and among the national parks across the United States.

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APPENDIX A.

All species data used in the analysis. Each species is listed with AOU number, genus, species, and common name. The most highly resolved (largest) analysis category for each species within each park is listed, with parks indicated by four-leeter codes. Analysis categories are nested (see text) so that a code 5 also implies codes 4, 3, 2, and 1 for an indicated species and park. Blank entries indicate that the species was not listed as occurring within the park.

Analysis Category Definitions:

1 = total species (no additional information given)
2 = regularly occurring species
3 = full-time resident and nesting migratory species
4 = nesting species
5 = full-time resident nesting species

(Note: Cabrillo NM (CABR) did not provide information on residency status and, thus does not include any category 3 species despite having some category 4 species.)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
GAVIIDAE																		
11.0	Gavia stellata	Red-throated Loon	2	2			1						3	2	2			
10.0	Gavia pacifica	Arctic Loon	2	2			1						3	2	2			1
7.0	Gavia immer	Common Loon	2	2		1	1	2	2	2			3	2	2		2	1
8.0	Gavia adamsii	Yellow-billed Loon					1						1					
PODICIPEDIDAE																		
6.0	Podilymbus podiceps	Pied-billed Grebe		1		3	1	3	3	5		1	5	3	5	1	2	3
3.0	Podiceps auritus	Horned Grebe	2	2		2	1		2				2	2	2		2	
2.0	Podiceps grisegena	Red-necked Grebe				1	1		2				2	2	2			
4.0	Podiceps nigricollis	Eared Grebe	2	2		2	1	2	2	2			2	2	2	4	2	2
1.0	Aechmophorus occidentalis	Western Grebe	2	2		1	1		3	2		1	3	2	2		2	1
DIOMEDEIDAE																		
81.0	Diomedea nigripes	Black-footed Albatross	1	1			1						2	2				
82.1	Diomedea immutabilis	Laysan Albatross											1	2				

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
PROCELLARIIDAE																		
86.0	Fulmarus glacialis	Northern Fulmar	2	2			1						2	2				
99.0	Pterodroma inexpectata	Mottled Petrel											1					
91.0	Puffinus creatopus	Pink-footed Shearwater		2			1						2	2				
95.1	Puffinus carneipes	Flesh-footed Shearwater					1						1					
96.2	Puffinus bulleri	Buller's Shearwater					1						1	2				
95.0	Puffinus griseus	Sooty Shearwater	2	2			1						2	2				
96.0	Puffinus tenuirostris	Short-tailed Shearwater	1				1						1					
90.0	Puffinus puffinus	Manx Shearwater	2				1						1					
93.0	Puffinus opisthomelas	Black-vented Shearwater		1														
HYDROBATIDAE																		
109.0	Oceanites oceanicus	Wilson's Storm-petrel					1											
105.0	Oceanodroma furcata	Fork-tailed Storm-petrel					1						1	2				
106.0	Oceanodroma leucorhoa	Leach's Storm-petrel		4			1						1	2				
108.0	Oceanodroma homochroa	Ashy Storm-petrel		4			1						4					
107.0	Oceanodroma melania	Black Storm-petrel		4									1					
103.0	Oceanodroma microsoma	Least Storm-petrel		1														
PHAETHONTIDAE																		
113.0	Phaethon aethereus	Red-billed Tropicbird		1														
SULIDAE																		
114.1	Sula nebouxii	Blue-footed Booby	1															
115.0	Sula leucogaster	Brown Booby		1														
PELECANIDAE																		
125.0	Pelecanus erythrorhynchos	American White Pelican		1		1	1	2	2	1			1	1	2			1
126.0	Pelecanus occidentalis	Brown Pelican	2	5			1	1					3	2	5			
PHALACROCORACIDAE																		
120.0	Phalacrocorax auritus	Double-crested Cormorant	2	5		2	1		2	2			3	3	5		2	1
122.0	Phalacrocorax pencillatus	Brandt's Cormorant	2	5			1						5	2	5			
123.0	Phalacrocorax pelagicus	Pelagic Cormorant	2	5			1						5	3	2			

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
FREGATIDAE																		
128.0	Fregata magnificens	Magnificent Frigatebird	2	1			1						1					
ARDEIDAE																		
190.0	Botaurus lentiginosus	American Bittern				2	1	2	3	1			4	3	2			
191.0	Ixobrychus exilis	Least Bittern				1	1		2				1		1			
194.0	Ardea herodias	Great Blue Heron	4	2		3	1	2	3	2		1	5	3	5	5	3	3
196.0	Casmerodius albus	Great Egret	2	1		2	1	2	3	1			5	2	2		1	1
197.0	Egretta thula	Snowy Egret	2	1		2	1	2	2	1			5	2	2	1		1
200.0	Egretta caerulea	Little Blue Heron				1							1					
199.0	Egretta tricolor	Tricolored Heron													2			
200.1	Bubulcus ibis	Cattle Egret	2	2		2	1						2	2	2			
201.0	Butorides striatus	Green-backed Heron		2		3	1	2	2	1		1	5	3	5		1	1
202.0	Nycticorax nycticorax	Black-crowned Night-heron	4	1		3	1	2	3	2			5	1	5	1		
203.0	Nycticorax violaceus	Yellow-crowned Night-heron											1					
THRESKIORNITHIDAE																		
184.0	Eudocimus albus	White Ibis					1						1					
187.0	Plegadis chihi	White-faced Ibis				1		2	2				1		2			
183.0	Ajaia ajaja	Roseate Spoonbill													1			
CICONIIDAE																		
188.0	Mycteria americana	Wood Stork				1												
ANATIDAE																		
178.0	Dendrocygna bicolor	Fulvous Whistling-duck				1												
180.0	Cygnus columbianus	Tundra Swan				1	1		2	2			2	2			1	1
181.0	Cygnus buccinator	Trumpeter Swan					1						2					
171.0	Anser albifrons	Greater White-fronted Goose		2		1	1		2				2	2			1	
169.0	Chen caerulescens	Snow Goose		2		1	1	2	2	2			1	2	2		1	1
170.0	Chen rossii	Ross' Goose				1	1		2				1	1			1	
176.0	Chen canagica	Emperor Goose					1		2				1	1				
173.0	Branta bernicla	Brant	2	2			1	1					2	2	2			
172.0	Branta canadensis	Canada Goose		2		2	1	2	3	3			2	2	2	1	2	1

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
144.0	Aix sponsa	Wood Duck		1		2	1		2	4			5	3	1	1	2	4
139.0	Anas crecca	Green-winged Teal		2		3	1	2	3	2			2	2	2	1	2	1
132.0	Anas platyrhynchos	Mallard		2	1	3	1	5	3	5	3	1	5	3	5	5	3	4
143.0	Anas acuta	Northern Pintail		2		3	1		3	2			5	2	2	1	1	1
140.0	Anas discors	Blue-winged Teal		2		2	1	2	2	1			2	2	2			1
141.0	Anas cyanoptera	Cinnamon Teal		2		3	1	2	2	1			5	3	2		1	1
142.0	Anas clypeata	Northern Shoveler		1		2	1		3	2			2	2	2	1	1	1
135.0	Anas strepera	Gadwall				2	1		3				5	3	2		1	1
136.0	Anas penelope	Eurasian Wigeon				1	1						2	1				
137.0	Anas americana	American Wigeon		1		2	1		2	2			2	2	2	1	2	1
147.0	Aythya valisineria	Canvasback				2	1	2	3	2			2	2	2	1	2	1
146.0	Aythya americana	Redhead				2	1		3				2	2	2	1	1	
150.0	Aythya collaris	Ring-necked Duck				2	1	2	2	4			2	2	2		2	1
149.1	Aythya fuligula	Tufted Duck											1					
148.0	Aythya marila	Greater Scaup				1	1		2				2	2	2		2	
149.0	Aythya affinis	Lesser Scaup		1		2	1	2	3	2			2	2	2	1	2	1
162.0	Somateria spectabilis	King Eider					1						1					
155.0	Histrionicus histrionicus	Harlequin Duck		1			1			1			1	2		1		4
154.0	Clangula hyemalis	Oldsquaw					1						2	2	2			
163.0	Melanitta nigra	Black Scoter		1			1						2	2	2			
166.0	Melanitta perspicillata	Surf Scoter	2	3		1	1						3	2	2		1	
165.0	Melanitta fusca	White-winged Scoter	2	2		1	1		2				3	2	2			
151.0	Bucephala clangula	Common Goldeneye		1		1	1		2	1			2	2	2		2	1
152.0	Bucephala islandica	Barrow's Goldeneye				1	1			1			2	1			2	1
153.0	Bucephala albeola	Bufflehead	2			2	1	2	2	5			2	2	2	1	2	1
131.0	Lophodytes cucullatus	Hooded Merganser				1	1		2				2	2	2		2	1
129.0	Mergus merganser	Common Merganser				1	1	2	3	4			1	2	2	4	2	5
130.0	Mergus serrator	Red-breasted Merganser	2	2		2	1	1					3	2	2		2	
167.0	Oxyura jamaicensis	Ruddy Duck		2		3	1	2	3	2			5	2	5	1	2	1

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
CATHARTIDAE																		
325.0	Cathartes aura	Turkey Vulture				2	1	3	2	2	3	5	5	2	5	4	2	2
324.0	Gymnogyps californianus	California Condor										1				1		
ACCIPITRIDAE																		
364.0	Pandion haliaetus	Osprey	2	2		1	1	2	2	2			5	2	2	1	2	1
328.0	Elanus caeruleus	Black-shouldered Kite		1			1	1				1	5	3	5	1		1
329.0	Ictinia mississippiensis	Mississippi Kite				1							1					
352.0	Haliaeetus leucocephalus	Bald Eagle	1	2		1	1	1	3	5			1	2		1	3	2
331.0	Circus cyaneus	Northern Harrier	2	2		2	1	2	3	2		1	5	3	2	1		2
332.0	Accipiter striatus	Sharp-shinned Hawk	2	2		2	1	2	3	4		5	4	2	2	5	2	5
333.0	Accipiter cooperii	Cooper's Hawk	2	2		2	1	2	3	4		5	5	3	5	5	3	5
334.0	Accipiter gentilis	Northern Goshawk			1	1			3	5			1		1	5	1	5
339.0	Buteo lineatus	Red-shouldered Hawk				1	1					1	5	2	5	5		1
343.0	Buteo platypterus	Broad-winged Hawk	1			1	1						1					
342.0	Buteo swainsoni	Swainson's Hawk	1	1		1	1	1	2			1	1			1		1
340.0	Buteo albonotatus	Zone-tailed Hawk	1															
337.0	Buteo jamaicensis	Red-tailed Hawk	2	5	1	3	1	5	3	3	3	5	5	3	5	5	3	5
348.0	Buteo regalis	Ferruginous Hawk	2			1	1	2	2	1		1	2		1			
347.0	Buteo lagopus	Rough-legged Hawk		1		1	1	1	2	1			2	2		1		1
349.0	Aquila chrysaetos	Golden Eagle	2	2		2	1	5	3	3		5	5	1	5	5	3	5
FALCONIDAE																		
360.0	Falco sparverius	American Kestrel	4	5		3	1	5	3	2	3	5	5	3	5	5	3	5
357.0	Falco columbarius	Merlin	2	2		1	1	2	2	1		1	2	2	2	1	2	1
356.0	Falco peregrinus	Peregrine Falcon	2	2		1	1	1	3	2		1	2	2	2	1	2	5
355.0	Falco mexicanus	Prairie Falcon	2	1		3	1	5	3	4		5	2	1	5	2		4
PHASIANIDAE																		
288.2	Alectoris chukar	Chukar				3		1	3			1				3		
309.1	Phasianus colchicus	Ring-necked Pheasant					1		3				5	1	2			
309.5	Pavo cristatus	Common Peafowl		5														
297.0	Dendragapus obscurus	Blue Grouse			1				3	4				3		5		5

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
304.0	Lagopus leucurus	White-tailed Ptarmigan																5
300.0	Bonasa umbellus	Ruffed Grouse							3	1				3				
309.0	Centrocercus urophasianus	Sage Grouse							3									1
310.0	Meleagris gallopavo	Wild Turkey		5						1		1				3	3	
295.0	Callipepla gambelii	Gambel's Quail				3		5										
294.0	Callipepla californica	California Quail	4	5		3	1		3	2	3	5	5	3	5	5	3	1
292.0	Oreortyx pictus	Mountain Quail			1	3		5	3	5				3	5	5	3	5
RALLIDAE																		
215.0	Coturnicops noveboracensis	Yellow Rail					1		1									1
216.0	Laterallus jamaicensis	Black Rail					1						5					
211.0	Rallus longirostris	Clapper Rail					1						5					
212.0	Rallus limicola	Virginia Rail		2		2	1	2	3	1		1	5	3	5			4
214.0	Porzana carolina	Sora		2		2	1	2	2	1			2	3	2	1		1
218.0	Porphyrula martinica	Purple Gallinule	1															
219.0	Gallinula chloropus	Common Moorhen				2	1						2	1	2			
221.0	Fulica americana	American Coot	2	2		3	1	5	3	5		1	5	2	5	1	2	1
GRUIDAE																		
206.0	Grus canadensis	Sandhill Crane				1	1		2	2			1					
CHARADRIIDAE																		
270.0	Pluvialis squatarola	Black-bellied Plover	2	3		1	1	1	2				3	2	2			
272.0	Pluvialis dominica	Lesser Golden-plover		2			1						2	2				
278.0	Charadrius alexandrinus	Snowy Plover		5		1	1		2				5	3				
274.0	Charadrius semipalmatus	Semipalmated Plover	2	2		1	1			1			2	2	2			
273.0	Charadrius vociferus	Killdeer	2	5		1	1	2	3	4		4	5	3	5	5	3	4
281.0	Charadrius montanus	Mountain Plover		2		2	1		1				1					
HAEMATOPODIDAE																		
286.0	Haematopus palliatus	American Oystercatcher	1	1														
287.0	Haematopus bachmani	American Black Oystercatcher	1	5			1						5	3	5			

APPENDIX A. (continued)

RECURVIROSTRIDAE

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
226.0	Himantopus mexicanus	Black-necked Stilt				2	1	2	2				4	1	4			1
225.0	Recurvirostra americana	American Avocet		1		2	1	2	2				2	2	5	1		1

SCOLOPACIDAE

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
254.0	Tringa melanoleuca	Greater Yellowlegs	2	2		2	1	2	2	2			2	2	2		1	1
255.0	Tringa flavipes	Lesser Yellowlegs		1		2	1		2			1	2	2	2			
256.0	Tringa solitaria	Solitary Sandpiper		2		1	1	2	2				1	1	2	1		1
258.0	Catoptrophorus semipalmatus	Willet	2	3		1	1		2	1			3	2	5			1
259.0	Heteroscelus incanus	Wandering Tattler	2	3			1						2	2	2			
263.0	Actitis macularia	Spotted Sandpiper	2	2	1	2	1	2	2	4		1	2	3	2	4	3	4
261.0	Bartramia longicauda	Upland Sandpiper		1		1			1					1				
265.0	Numenius phaeopus	Whimbrel	2	3		1	1						3	2	2			
264.0	Numenius americanus	Long-billed Curlew	2	2		1	1	2	2				3	2	2	1		
251.0	Limosa haemastica	Hudsonian Godwit												1				
250.0	Limosa lapponica	Bar-tailed Godwit											1					
249.0	Limosa fedoa	Marbled Godwit	2	3		1	1	1	2				3	2	2			
283.0	Arenaria interpres	Ruddy Turnstone	2	3		1	1						2	2	2			
284.0	Arenaria melanocephala	Black Turnstone	2	3			1						2	2	2			
282.0	Aphriza virgata	Surfbird	2	2			1						2	2	2			
234.0	Calidris canutus	Red Knot		1		1	1						2	2	2			
248.0	Calidris alba	Sanderling	2	3		1	1	1					2	2	5			
245.0	Calidris pusilla	Semipalmated Sandpiper											1					
246.0	Calidris mauri	Western Sandpiper		2		2	1	2	2	2			2	2	5			
242.2	Calidris ruficollis	Rufous-necked Stint												1				
242.0	Calidris minutilla	Least Sandpiper	2	2		2	1	2	2	2		1	2	2	2			
240.0	Calidris fuscicollis	White-rumped Sandpiper											1					
241.0	Calidris bairdii	Baird's Sandpiper		1		1	1		2				2	2	2		1	
239.0	Calidris melanotos	Pectoral Sandpiper		2		1	1						2	2	2			
238.0	Calidris acuminata	Sharp-tailed Sandpiper					1						1	1				
236.0	Calidris ptilocnemis	Rock Sandpiper					1						1	2				
243.0	Calidris alpina	Dunlin		2		1	1		2				2	2	2			
244.0	Calidris ferruginea	Curlew Sandpiper											1					

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
233.0	Calidris himantopus	Stilt Sandpiper				1	1						1	1				
262.0	Tryngites subruficollis	Buff-breasted Sandpiper											1					
260.0	Philomachus pugnax	Ruff											1	1				
231.0	Limnodromus griseus	Short-billed Dowitcher		1		2	1						2	2	2			
232.0	Limnodromus scolopaceus	Long-billed Dowitcher		2		2	1	2	2				2	2	2			
230.0	Gallinago gallinago	Common Snipe		2		2	1	2	3	4		1	2	2	2		2	1
224.0	Phalaropus tricolor	Wilson's Phalarope		1		2	1	2	2	1			2	2	2		1	1
223.0	Phalaropus lobatus	Red-necked Phalarope	2	2		2	1	2	2	2			2	2	2	1	2	1
222.0	Phalaropus fulicaria	Red Phalarope	2	2		1	1						2	2	2			
LARIDAE																		
36.0	Stercorarius pomarinus	Pomarine Jaeger		1			1						2	2				
37.0	Stercorarius parasiticus	Parasitic Jaeger	2	2			1						2	2				
38.0	Stercorarius longicaudus	Long-tailed Jaeger											1					
35.2	Catharcta maccormicki	South Polar Skua											1					
59.0	Larus pipixcan	Franklin's Gull		1		1	1						1	1	1			
55.1	Larus ridibundus	Common Black-headed Gull											1					
60.0	Larus philadelphia	Bonaparte's Gull	2	2		1	1		2				2	2	2		2	
57.0	Larus heermanni	Heermann's Gull	2	3			1						2	2	2			
55.0	Larus canus	Mew Gull	2	2			1						2	2	2			
54.0	Larus delawarensis	Ring-billed Gull	2	1		2	1	2	3	2			3	2	5	1	2	1
53.0	Larus californicus	California Gull	2	2	1	2	1	2	3	2			3	2	5	2	2	2
51.0	Larus argentatus	Herring Gull	2	2			1		2				2	2	2		2	
43.1	Larus thayeri	Thayer's Gull		1			1						2	2	2		1	
49.0	Larus occidentalis	Western Gull	2	5			1						5	3	5			
44.0	Larus glaucescens	Glaucous-winged Gull	2	1			1						3	2	2		1	
42.0	Larus hyperboreus	Glaucous Gull	1	2			1						2	1				
40.0	Rissa tridactyla	Black-legged Kittiwake	2	2			1						2	2	2			
62.0	Xema sabini	Sabine's Gull				1	1			1			2	2				
64.0	Sterna caspia	Caspian Tern	2				1		2	2			2	2	2		1	1
65.0	Sterna maxima	Royal Tern	2	2			1						1		2			
66.0	Sterna elegans	Elegant Tern	2				1						2	1	2			
70.0	Sterna hirundo	Common Tern	2				1						2	2	2			

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
71.0	Sterna paradisaea	Arctic Tern		1			1						1		2			
69.0	Sterna forsteri	Forster's Tern	2	2		1	1		2				3	2	5		1	
74.0	Sterna antillarum	Least Tern	2			1	1						1	1	4			
77.0	Chlidonias niger	Black Tern	1			1	1	2	2				1	1	2			
ALCIDAE																		
30.0	Uria aalge	Common Murre	2	2			1						5	3	2			
29.0	Cepphus columba	Pigeon Guillemot		4			1						5	3	2			
23.0	Brachyramphus marmoratus	Marbled Murrelet					1						2	3				
25.0	Synthliboramphus hypoleucus	Xantus' Murrelet	2	4									1					
26.0	Synthliboramphus craveri	Craveri's Murrelet		1														
21.0	Synthliboramphus antiquus	Ancient Murrelet					1						2	1				
16.0	Ptychoramphus aleuticus	Cassin's Auklet		5			1						3	3				
17.0	Cyclorrhynchus psittacula	Parakeet Auklet					1						1					
18.0	Aethia cristatella	Crested Auklet											1					
15.0	Cerorhinca monocerata	Rhinoceros Auklet	2	2			1						3	3				
12.0	Fratercula cirrhata	Tufted Puffin		1			1						2	2				
14.0	Fratercula corniculata	Horned Puffin		1			1						1					
COLUMBIDAE																		
313.1	Columba livia	Rock Dove	2	2		3	1	3	3	1		3	5	3	2	3	1	
312.0	Columba fasciata	Band-tailed Pigeon	2	2		1	1	2	3	2	3	3	5	2	5	5	3	4
315.2	Streptopelia risoria	Ringed Turtle-dove		1											2			
315.1	Streptopelia chinensis	Spotted Dove		1											2			
319.0	Zenaida asiatica	White-winged Dove	2	1		1	1	2				1	1		1			
316.0	Zenaida macroura	Mourning Dove	4	5		3	1	5	3	2	3	4	5	2	5	5	2	4
321.0	Columbina inca	Inca Dove				1												
320.0	Columbina passerina	Common Ground-dove				1		2										
CULCULIDAE																		
388.0	Coccyzus erythropthalmus	Black-billed Cuckoo					1						1					
387.0	Coccyzus americanus	Yellow-billed Cuckoo				1	1	1					1	1				
385.0	Geococcyx californianus	Greater Roadrunner				3		5				3			5	1	3	1

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
TYTONIDAE																		
365.0	Tyto alba	Common Barn-owl	2	5		3	1	5	3	1		5	5	3	5	1		1
STRIGIDAE																		
374.0	Otus flammeolus	Flammulated Owl		1		1				1						4	1	4
373.2	Otus kennicottii	Western Screech-owl				3	1	5	3	1		2	5	3	5	5	3	3
375.0	Bubo virginianus	Great Horned Owl	2	1	1	3	1	5	3	5	3	5	5	3	5	5	3	5
376.0	Nyctea scandiaca	Snowy Owl					1						1	1				
379.0	Glaucidium gnoma	Northern Pygmy-owl				1	1		3	3		1	3	3		5	3	5
378.0	Athene cunicularia	Burrowing Owl	2	3		1	1	3	3				2	2	5	1		
369.0	Strix occidentalis	Spotted Owl			1		1			1	3		5	3		5		5
370.0	Strix nebulosa	Great Gray Owl			1				3	1			1			1		5
366.0	Asio otus	Long-eared Owl	1	2	1	1	1	2	3	1		1	2			1	1	4
367.0	Asio flammeus	Short-eared Owl		5		1	1		3	1			2	3	2	1		1
372.0	Aegolius acadicus	Northern Saw-whet Owl		5		1	1	1	3	1	3	1	5	3		5	1	5
CAPRIMULGIDAE																		
421.0	Chordeiles acutipennis	Lesser Nighthawk	2	2		2	1	4					1		2			
420.0	Chordeiles minor	Common Nighthawk		1	1	1			2	2			1	2		4	1	2
418.0	Phalaenoptilus nuttallii	Common Poorwill	2	2		3	1	5	2	4		3	4		4	4	2	2
417.0	Caprimulgus vociferus	Whip-poor-will	1															
APODIDAE																		
422.0	Cypseloides niger	Black Swift	1	1	1	1	1						2	1	2	4	1	4
423.0	Chaetura pelagica	Chimney Swift		2		1							1		4			
424.0	Chaetura vauxi	Vaux's Swift	2	2		2	1	2	2	2			2	2	2	4	1	2
425.0	Aeronautes saxatalis	White-throated Swift	2	5		3	1	5		2		5	5		5	5		4
TROCHILIDAE																		
427.0	Lampornis clemenciae	Blue-throated Hummingbird														1		
429.0	Archilochus alexandri	Black-chinned Hummingbird	2			2	1	2	1			1	1		4	4	1	2
431.0	Calypte anna	Anna's Hummingbird	4	5		1	1	5	2	2	3	5	5	2	5	5	3	2
430.0	Claypte costae	Costa's Hummingbird	2	2		2	1	5				2	1		4			
436.0	Stellula calliope	Calliope Hummingbird	2	1	1	2	1	2	2	4			1			4	1	4

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
432.0	Selasphorus platycercus	Broad-tailed Hummingbird				2			1									
433.0	Selasphorus rufus	Rufous Hummingbird	2	2		2	1	2	2	2		2	2	2	2	2	2	2
434.0	Selasphorus sasin	Allen's Hummingbird	2	5			1	2			2	4	4	2	4	2	1	1
ALCEDINIDAE																		
390.0	Ceryle alcyon	Belted Kingfisher	2	2	1	2	1	2	3	3	3	2	5	3	2	5	3	5
PICIDAE																		
408.0	Melanerpes lewis	Lewis' Woodpecker		2		1	1	1	1	2		1	1	1	2	1	2	2
407.0	Melanerpes formicivorus	Acorn Woodpecker	2	5		1	1	2		1	3	5	5	3	5	5	3	5
402.1	Sphyrapicus nuchalis	Red-naped Sapsucker	2	2	1	2	1	2	3	4	2	2	1		2	5	3	
403.0	Sphyrapicus ruber	Red-breasted Sapsucker		2		2		2					2	3	2			5
404.0	Sphyrapicus thyroideus	Williamson's Sapsucker				1		1	3	5			1			5	1	5
396.0	Picoides scalaris	Ladder-backed Woodpecker				3		5										
397.0	Picoides nuttallii	Nuttall's Woodpecker	1	1			1					5	5		5	5	3	5
394.0	Picoides pubescens	Downy Woodpecker				1	1		3	5	3	3	5	3	5	5	3	5
393.0	Picoides villosus	Hairy Woodpecker			1	3	1		3	5	3	5	5	3	2	5	3	5
399.0	Picoides albolarvatus	White-headed Woodpecker			1	1		1	3	5			1			5		5
400.0	Picoides arcticus	Black-backed Woodpecker			1				3	5						5		5
412.0	Colaptes auratus	Northern Flicker	2	5	1	3	1	1	3	5	3	5	5	3	5	5	3	5
405.0	Dryocopus pileatus	Pileated Woodpecker					1			5	3		5	3		5	3	5
TYRANNIDAE																		
459.0	Contopus borealis	Olive-sided Flycatcher	2	2	1	2	1	2	2	4	3	2	4	2	2	4	2	4
462.0	Contopus sordidulus	Western Wood-pewee	2	2	1	2	1	2	2	4		4	4	2	4	4	2	4
466.0	Empidonax traillii	Willow Flycatcher	2	2	1	2	1	2		4		2	2	2	2	4	1	4
467.0	Empidonax minimus	Least Flycatcher	1	1		1	1						1					
468.0	Empidonax hammondii	Hammond's Flycatcher	2	2		2	1	2	2	4			1	2	2	4	1	4
469.0	Empidonax oberholseri	Dusky Flycatcher	2	2		2	1	2	2	4			1	2	2	4	1	4
469.1	Empidonax wrightii	Gray Flycatcher	1	2		2	1	2	2				1			1		1
464.0	Empidonax difficilis	Western Flycatcher	2	5	1	2	1	2		2	2	4	4	2	4	4	2	4
458.0	Sayornis nigricans	Black Phoebe	2	5		3	1	5		1	3	5	5	3	5	5	3	5
456.0	Sayornis phoebe	Eastern Phoebe	1			1	1						1					

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
457.0	Sayornis saya	Say's Phoebe	2	2		3	1	5	2			5	2		2	1		4
471.0	Pyrocephalus rubinus	Vermilion Flycatcher				2		1										
455.0	Myiarchus tuberculifer	Dusky-capped Flycatcher				1												
454.0	Myiarchus cinerascens	Ash-throated Flycatcher	2	4		2	1	4	2	1		4	4	2	4	4	2	1
452.0	Myiarchus crinithus	Great Crested Flycatcher	1										1					
453.0	Myiarchus tyrannulus	Brown-crested Flycatcher				1												
446.0	Tyrannus melancholicus	Tropical Kingbird	1				1						2	1				
448.0	Tyrannus vociferans	Cassin's Kingbird	2	2		1	1	2					1		4			
455.1	Tyrannus crassirostris	Thick-billed Kingbird	1															
447.0	Tyrannus verticalis	Western Kingbird	2	2		2	1	4	2	1		4	4	2	4	4	1	2
444.0	Tyrannus tyrannus	Eastern Kingbird	1	1		1	1						1	1				
443.0	Tyrannus forficatus	Scissor-tailed Flycatcher		1		1	1						1					
ALAUDIDAE																		
473.0	Alauda arvensis	Eurasian Skylark											1					
474.0	Eremophila alpestris	Horned Lark	2	5		3	1	5	3	2			5	2	5	4		4
HIRUNDINIDAE																		
611.0	Progne subis	Purple Martin	2	1		1	1	2	2	1			4	2	2	1	1	
614.0	Tachycineta bicolor	Tree Swallow	2	2	1	2	1	2	2	4		1	5	2	2	4	2	4
615.0	Tachycineta thalassina	Violet-green Swallow		2	1	2	1	2	2	2	3	4	4	2	4	4	2	4
617.0	Stelgidopteryx serripennis	Northern Rough-winged Swallow	2	2		2	1	2	2	2		2	4	2	4	4	2	4
616.0	Riparia riparia	Bank Swallow		1		2	1	2	2				1		2		1	
612.0	Hirundo pyrrhonota	Cliff Swallow	2	2		2	1	2	2	2		4	4	2	4	4	1	4
613.0	Hirundo rustica	Barn Swallow	4	4		2	1	2	2	2		1	4	2	4	1	2	4
CORVIDAE																		
484.0	Perisoreus canadensis	Gray Jay								5				3				
478.0	Cyanocitta stelleri	Stellar's Jay	2		1	1	1	1	3	5	3	2	5	3		5	3	5
477.0	Cyanocitta cristata	Blue Jay				1							1	1				
481.0	Aphelocoma coerulescens	Scrub Jay	4	5		3	1	5	3	1	3	5	5	3	5	5	3	3
492.0	Gymnorhinus cyanocephalus	Pinyon Jay				3		5	1				1			1		2
491.0	Nucifraga columbiana	Clark's Nutcracker	1	1	1	3	1	1		5			1	1		5	1	5

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APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
475.0	Pica pica	Black-billed Magpie				1			3									1
476.0	Pica nuttalli	Yellow-billed Magpie					1					2	1					1
488.0	Corvus brachyrhynchos	American Crow		1		2	1			2	3	2	5	3	5	2	3	2
486.0	Corvus corax	Common Raven	4	5	1	3	1	5	3	3	3	5	5	3	5	5	3	3
PARIDAE																		
735.0	Parus atricapillus	Black-capped Chickadee												3				
738.0	Parus gambeli	Mountain Chickadee	2		1	3	1	1	3	5			1	1	2	5	3	5
741.0	Parus rufescens	Chestnut-backed Ckickadee					1				3	3	5	3			3	5
733.0	Parus inornatus	Plain Titmouse	2			3	1	5	3	1		5	5		5	5	3	5
REMIZIDAE																		
746.0	Auriparus flaviceps	Verdin				1		5										
AEGITHALIDAE																		
743.0	Psaltriparus minimus	Bushtit	4	5		3	1	5	3	2	3	5	5	3	5	5	3	5
SITTIDAE																		
728.0	Sitta canadensis	Red-breasted Nuthatch	4	5	1	2	1	2	3	5		1	5	2	2	5	3	5
727.0	Sitta carolinensis	White-breasted Nuthatch	2	2	1	3	1	2	3	5		5	5	1	5	5	3	5
730.0	Sitta pygmaea	Pygmy Nuthatch	2			1	1	1	3	5	3		5			5	2	5
CERTHIIDAE																		
726.0	Certhia americana	Brown Creeper	2	2	1	2	1	2	3	5	3	1	5	3	2	5	3	5
TROGLODYTIDAE																		
713.0	Campylorhynchus brunneicapillus	Cactus Wren				1		5							5			
715.0	Salpinctes obsoletus	Rock Wren	2	5	1	3	1	5	2			5	5		5	5	1	5
717.0	Catherpes mexicanus	Canyon Wren		3	1	3		5	3	1		5			5	5	1	5
719.0	Thryomanes bewickii	Bewick's Wren	4	5		3	1	5	3	1	3	5	5	3	5	5	3	4
721.0	Troglodytes aedon	House Wren	2	2	1	2	1	2	2	2		4	4	2	5	4	2	4
722.0	Troglodytes troglodytes	Winter Wren	1	2		2	1	2	2	3	3	1	5	3	2	5	2	5
724.0	Cistothorus platensis	Sedge Wren											1					
725.0	Cistothorus palustris	Marsh Wren		2		3	1	2	2				5	3	5	1		2

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
CINCLIDAE																		
701.0	Cinclus mexicanus	American Dipper		1	1	1	1			5		1	1	3	5	5	3	5
MUSCICAPIDAE																		
748.0	Regulus satrapa	Golden-crowned Kinglet	2	2	1	1	1	1	3	5	3	2	5	3	2	5	2	5
749.0	Regulus calendula	Ruby-crowned Kinglet	2	2	1	3	1	2	3	4	3	2	2	2	2	5	2	5
751.0	Polioptila caerulea	Blue-gray Gnatcatcher	2	5		2	1	5	2			4	1	1	5	5	2	4
752.0	Polioptila melanura	Black-tailed Gnatcatcher				3		5										
767.0	Sialia mexicana	Western Bluebird	2	1		2	1	2	3	5	3	5	5	2	5	5	3	5
768.0	Sialia currucoides	Mountain Bluebird		2	1	3	1	1	3	5			1	1		4		4
754.0	Myadestes townsendi	Townsend's Solitaire	1	2	1	3	1	1	3	5		1	1	3	2	5	3	3
756.0	Catharus fuscescens	Veery											1					
757.0	Catharus minimus	Gray-cheeked Thrush											1					
758.0	Catharus ustulatus	Swainson's Thrush	2	2		1	1	2		4	2	2	4	2	4	4		4
759.0	Catharus guttatus	Hermit Thrush	2	2	1	3	1	2	3	4	3	2	5	2	2	5	2	5
755.0	Hylocichla mustelina	Wood Thrush											1					
762.2	Turdus rufopalliatus	Rufous-backed Robin				1												
761.0	Turdus migratorious	American Robin	4	2	1	3	1	1	3	5	3	2	5	3	5	5	3	5
763.0	Ixoreus naevius	Varied Thrush	1	2		1	1	1	2	2	2	2	2	3	2	1	2	1
742.0	Chamaea fasciata	Wrentit	4				1			1		5	5	3	5	5	3	3
MIMIDAE																		
704.0	Dumetella carolinensis	Gray Catbird	1	1		1							1					
703.0	Mimus polyglottos	Norhern Mockingbird	4	5		3	1	5				1	5	3	5	3		1
702.0	Oreoscoptes montanus	Sage Thrasher	2	2		2	1	1	2				1					1
705.0	Toxostoma rufum	Brown Thrasher	1			1	1	1					1					
708.0	Toxostoma bendirei	Bendire's Thrasher		1		1		2										
710.0	Toxostoma redivivum	California Thrasher	4				1	5				5	3		5	5	3	
712.0	Toxostoma dorsale	Crissal Thrasher				1												
711.0	Toxostoma lecontei	Le Conte's Thrasher				3		5										
MOTACILLIDAE																		
696.0	Motacilla flava	Yellow Wagtail											1					
694.0	Motacilla alba	White Wagtail											1					

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APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
699.0	Anthus cervinus	Red-throated Pipit		1			1						1					
697.0	Anthus spinoletta	Water Pipit	2	2	1	2	1	1	2	2			2	2	2	4	1	4
700.0	Anthus spragueii	Sprague's Pipit				1												
BOMBYCILLIDAE																		
618.0	Bombycilla garrulus	Bohemian Waxwing				1			1									1
619.0	Bombycilla cedrorum	Cedar Waxwing	2	2		2	1	1	2	2		2	2	2	2	2	2	3
PTILOGONATIDAE																		
620.0	Phainopepla nitens	Phainopepla	2	2		2	1	5				5	1		5	1		1
LANIIDAE																		
621.0	Lanius excubitor	Northern Shrike				1	1		1				1	2				1
622.0	Lanius ludovicianus	Loggerhead Shrike	4	5		3	1	5	3			2	5	2	5	1		1
STURNIDAE																		
493.0	Sturnus vulgaris	European Starling	4	5		3	1	5	3	1		3	5	3	2	5	3	5
VIREONIDAE																		
631.0	Vireo griseus	White-eyed Vireo					1						1					
633.0	Vireo bellii	Bell's Vireo	1			1		1				1			1			
634.0	Vireo vicinior	Gray Vireo		1		2		4				1						
629.0	Vireo solitarius	Solitary Vireo	2	2		2	1	2	2	4		4	4	2		4	2	4
628.0	Vireo flavifrons	Yellow-throated Vireo				1							1					
632.0	Vireo huttoni	Hutton's Vireo		5	1	1	1	1		1	3	3	5	2	5	5	3	5
627.0	Vireo gilvus	Warbling Vireo	2	2	1	2	1	2	2	4	2	4	4	2	4	4	2	4
626.0	Vireo philedelphicus	Philadelphia Vireo	1	1		1	1						1					
624.0	Vireo olivaceus	Red-eyed Vireo	1	1		1	1						1					
EMBERIZIDAE																		
641.0	Vermivora pinus	Blue-winged Warbler	1															
642.0	Vermivora chrysoptera	Golden-winged Warbler				1												
647.0	Vermivora peregrina	Tennessee Warbler	2	2		2	1	2					2		1			1
646.0	Vermivora celata	Orange-crowned Warbler	4	5		2	1	2	2	2		4	5	2	5	4	2	2

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APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
645.0	Vermivora ruficapilla	Nashville Warbler	2	2		2	1	2	2	4		2	1	2	2	4	2	4
644.0	Vermivora virginiae	Virginia's Warbler	1	1		1	1	2					1		1			1
643.0	Vermivora luciae	Lucy's Warbler	1	1		2	1	2					1					
648.0	Parula americana	Northern Parula	1	1		1	1						1					
652.0	Dendroica petechia	Yellow Warbler	2	2	1	2	1	2	2	4	2	2	4	2	4	4	2	4
659.0	Dendroica pensylvanica	Chestnut-sided Warbler	1	1		1	1						2					
657.0	Dendroica magnolia	Magnolia Warbler	2	2		1	1						2					1
650.0	Dendroica tigrina	Cape May Warbler	1	1		1	1						1					
654.0	Dendroica caerulescens	Black-throated Blue Warbler	1	1		1	1	1					2					
655.0	Dendroica coronata	Yellow-rumped Warbler	2	2	1	3	1	2	3	4		2	5	2	2	5	2	5
665.0	Dendroica nigrescens	Black-throated Gray Warbler	2	2		2	1	2	2	4		2	5	2	4	4	2	4
668.0	Dendroica townsendi	Townsend's Warbler	2	2		2	1	2	2	2	2	2	2	2	2	2	2	2
669.0	Dendroica occidentalis	Hermit Warbler	2	2	1	2	1	2	2	4		2	5	2	2	4	2	4
667.0	Dendroica virens	Black-throated Green Warbler	1	1		1	1						1	1				
662.0	Dendroica fusca	Blackburnian Warbler	1	1		1	1						2					
663.0	Dendroica dominica	Yellow-throated Warbler	2			1							1					
664.0	Dendroica graciae	Grace's Warbler	1															
671.0	Dendroica pinus	Pine Warbler	1				1						1					
673.0	Dendroica discolor	Prairie Warbler	1			1	1						1					
672.0	Dendroica palmarum	Palm Warbler		2		1	1	1					2	2				
660.0	Dendroica castanea	Bay-breasted Warbler	1	1		1	1						1					
661.0	Dendroica striata	Blackpoll Warbler	1	2		1	1						2					1
658.0	Denroica cerulea	Cerulean Warbler	1										1					
636.0	Mniotilta varia	Black-and-white Warbler	2	2		2	1	2					2	2	1	1		1
687.0	Setophaga ruticilla	American Redstart	2	2		2	1	2					2	2	1	1		
637.0	Protonotaria citrea	Prothonotary Warbler	1			1							1					
639.0	Helmitherus vermivorus	Worm-eating Warbler	1										1					
674.0	Seiurus aurocapillus	Ovenbird	1	2		1	1	1					1			1		
675.0	Seiurus noveboracensis	Northern Waterthrush	1	1		2	1	2				1	1					
677.0	Oporornis formosus	Kentucky Warbler	1	1														
678.0	Oporornis agilis	Connecticut Warbler	2	1		1	1						1					
679.0	Oporornis philadelphia	Mourning Warbler	1			1							1					

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
680.0	Oporornis tolmiei	MacGillivray's Warbler	2	2		2	1	2	2	4		1	4	2	2	4	2	4
681.0	Geothlypos trichas	Common Yellowthroat	2	2		2	1	2	2	1		1	5	2	5	1	2	2
684.0	Wilsonia citrina	Hooded Warbler	1			1							1					
685.0	Wilsonia pusilla	Wilson's Warbler	2	2	1	2	1	2	2	2	2	2	4	2	2	4	2	4
686.0	Wilsonia canadensis	Canada Warbler	1	1		1							1					
690.0	Cardellina rubrifrons	Red-faced Warbler	1															
688.0	Myioborus pictus	Painted Redstart	2	1		1		1										
683.0	Icteria virens	Yellow-breasted Chat	2	2		2	1	2				1	1	2			2	1
609.0	Piranga flava	Hepatic Tanager	1			1												
610.0	Piranga rubra	Summer Tanager	2	1		1		2					1					
608.0	Piranga olivacea	Scarlet Tanager	1			1	1						1					
607.0	Piranga ludoviciana	Western Tanager	2	2	1	2	1	2	2	4		2	4	2	2	4	2	4
595.0	Pheucticus ludovicianus	Rose-breasted Grosbeak	1	2		1	1						2					1
596.0	Pheucticus melanocephalus	Black-headed Grosbeak	2	4		2	1	2	2	4	2	4	4	2	4	4	2	4
597.0	Guiraca caerulea	Blue Grosbeak	2	2		2	1	2					1		4			1
599.0	Passerina amoena	Lazuli Bunting	2	2		2	1	2	2	4		2	4	2	4	4	2	4
598.0	Passerina cyanea	Indigo Bunting	1	2		1	1						1			1		
601.0	Passerina ciris	Painted Bunting	1			1												
604.0	Spiza americana	Dickcissel	1	1		1	1						1					
590.0	Pipilo chlorurus	Green-tailed Towhee	2	2		2	1	2	2	4		1	1			4		4
587.0	Pipilo erythrophthalmus	Rufous-sided Towhee	4	5		3	1	5	3	2	3	5	5	3	5	5	3	5
591.0	Pipilo fuscus	Brown Towhee	4				1	5		1	3	5	5		3	5	3	1
580.0	Aimophila ruficeps	Rufous-crowned Sparrow	4	5		1	1	1				5	5		5	5		1
559.0	Spizella arborea	American Tree Sparrow	1	1		2	1						1	1				
560.0	Spizella passerina	Chipping Sparrow	4	4		2	1	2	2	4		1	4	2	2	4	2	4
561.0	Spizella pallida	Clay-colored Sparrow	1	2		1	1						2	1				
562.0	Spizella breweri	Brewer's Sparrow	2	2		3	1	2	2	1			1			1		1
563.0	Spizella pusilla	Field Sparrow					1											
565.0	Spizella atrogularis	Black-chinned Sparrow	1	1		2	1	2				1	1		4	1		1
540.0	Pooecetes gramineus	Vesper Sparrow	1	2		2	1	2	2	1			1	2		1		1
552.0	Chondestes grammacus	Lark Sparrow	2	2		2	1	2	2	1		4	5		5	4	3	2
573.0	Amphispiza bilineata	Black-throated Sparrow	1	2		3	1	5	2				1			1		1

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APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
574.0	Amphispiza belli	Sage Sparrow	1	1		3	1	2	2			2	4	1	5	4	1	1
605.0	Calamospiza melancocorys	Lark Bunting		1		1	1	2					1					
542.0	Passerculus sandwichensis	Savannah Sparrow	2	2		3	1	2	2	1			5	3	5	1		2
546.0	Ammodramus savannarum	Grasshopper Sparrow		1		1	1	2					4	1	4			
548.0	Ammodramus leconteii	Le Conte's Sparrow				1												
549.0	Ammodracus caudacutus	Sharp-tailed Sparrow				1							1					
585.0	Passerella iliaca	Fox Sparrow	2	2	1	3	1	2	2	4	2	2	2	3	2	5	2	4
581.0	Melospiza melodia	Song Sparrow	2	5	1	3	1	2	3	4	3	3	5	3	5	5	3	4
583.0	Melospiza lincolnii	Lincoln's Sparrow	2	2	1	2	1	2		4		1	2	2	2	5	2	4
584.0	Melospiza georgiana	Swamp Sparrow		1		2	1						2	2				
558.0	Zonotrichia albicollis	White-throated Sparrow	2	1		2	1						2	2			1	1
557.0	Zonotrichia atricapilla	Golden-crowned Sparrow	2	2		2	1	2		2	2	2	2	2	2	2	2	2
554.0	Zonotrichia leucophrys	White-crowned Sparrow	2	2	1	2	1	2	2	4	3	2	5	3	2	5	2	4
553.0	Zonotrichia querula	Harris' Sparrow	1	1		2	1	2					1					1
567.0	Junco hyemalis	Dark-eyed Junco	2	2	1	3	1	2	3	5	3	5	5	3	5	5	1	5
539.0	Clacarius mccownii	McCown's Longspur		1									1					
536.0	Calcarius lapponicus	Lapland Longspur	1	1		1	1		1				1	2				
538.0	Calcarius ornatus	Chestnut-collared Longspur	1	1		2	1						1	1				
534.0	Plectrophenax nivalis	Snow Bunting				1							1	1				
494.0	Dolichonyx oryzivorus	Bobolink	1	2		2	1						1	1				1
498.0	Agelaius phoeniceus	Red-winged Blackbird		5	1	3	1	2	3	4		2	5	3	5	5	2	4
500.0	Agelaius tricolor	Tricolored Blackbird	1	2		1	1		3				5		5	1		
501.1	Sturnella neglecta	Western Meadowlark	2	5		3	1	2	3	2	3	2	5	3	5	2		4
497.0	Xanthocephalus xanthocephalus	Yellow-headed Blackbird	2	2		2	1	2	2	1			1	2	2	1		1
509.0	Euphagus carolinus	Rusty Blackbird	1	1		2	1						1	1				
510.0	Euphagus cyanocephalus	Brewer's Blackbird	4	2	1	3	1	2	3	4	3	2	5	3	5	5	3	4
512.0	Quiscalus mexicanus	Great-tailed Grackle				3		2										
511.0	Quiscalus quiscula	Common Grackle				1												
495.0	Molothrus ater	Brown-headed Cowbird	4	2	1	3	1	4	3	4		2	5	2	5	4	2	4
506.0	Icterus spurius	Orhcard Oriole	1			1	1						1					
505.0	Icterus cucullatus	Hooded Oriole	4	2		2	1	4					4	2	4			

APPENDIX A. (continued)

			CABR	CHIS	DEPO	DEVA	GOGA	JOTR	LABE	LAVO	MUWO	PINN	PORE	REDW	SAMO	SEKI	WHIS	YOSE
505.1	Icterus pustulatus	Streak-backed Oriole				1												
507.0	Icterus galbula	Northern Oriole	4	2		2	1	4	2	2		4	4	2	4	4	2	4
504.0	Icterus parisorum	Scott's Oriole	1	1		2		4					1					
FRINGILLIDAE																		
524.0	Leucosticte arctoa	Rosy Finch				1				5						5		5
515.0	Pinicola enucleator	Pine Grosbeak			1					1				1		5		4
517.0	Carpodacus purpureus	Purple Finch		2		2	1	2	3	2	3	3	5	3	5	5	2	4
518.0	Carpodacus cassinii	Cassin's Finch			1	3	1	1	3	4			1			5	1	5
519.0	Carpodacus mexicanus	House Finch	4	5		3	1	5	3	1	3	5	5	3	5	5		1
521.0	Loxia curvirostra	Red Crossbill	1	2	1	1	1	1	1	4			3	2		5		4
533.0	Carduelis pinus	Pine Siskin	2	2	1	2	1	1	3	5	3	2	5	3	2	5	2	4
530.0	Carduelis psaltria	Lesser Goldfinch	2	5		3	1	5	3	2		5	5	2	5	5	3	4
531.0	Carduelis lawrencei	Lawrence's Goldfinch	2	2		1	1	1		1		4	1		5	4		2
529.0	Carduelis tristis	American Goldfinch	2	2		3	1	1	3	2		1	5	3	5	1	1	2
514.0	Coccothraustes vespertinus	Evening Grosbeak			1	2	1		3	5			1	2		5	2	4
PASSERIDAE																		
688.2	Passer domesticus	House Sparrow	4	2		3	1	5				1	5	3	5	5		4

Monitoring of Golden Eagle Nests at Santa Monica Mountains National Recreation Area

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Abstract. Since 1983, the nesting status of golden eagles (*Aquila chrysaetos*) in the Santa Monica Mountains has been monitored by helicopter. The use of helicopters improved survey accuracy and efficiency, and was instrumental in the discovery of new nest sites. Results of the helicopter surveys indicate that individual nest site productivity declined from a ratio of 1.2 young per nest in 1982/1983 to 0.33 young per nest in 1988/1989. Environmental factors that may be influencing declining nest site productivity include habitat loss, human disturbance and drought.

INTRODUCTION

Shortly after the establishment of the Santa Monica Mountains National Recreation Area in 1978, the National Park Service embarked on an effort to inventory raptorial species in the park. Ornithologists, falconers, university professors, and local residents provided historical data, field notes and maps to the park. This information indicated that there were 13 species of raptors breeding in the Santa Monica Mountains. Historic records (Western Foundation for Vertebrate Zoology 1980) indicated that the California condor (*Gymnogyps californianus*) once nested in the Santa Monica Mountains; six peregrine falcon (*Falco peregrinus*) and three bald eagle (*Haliaeetus leucocephalus*) nest sites were also recorded in the park.

From this preliminary information, it became apparent that the Santa Monica Mountains provides important habitat for birds of prey, both in a historical context and in relation to current populations. Since golden eagles are sensitive to the quality of their environment and rare in the Santa Monica Mountains, it was determined that systematic surveys which monitored population status and nest site productivity were needed to develop protection strategies for the species, as well as to provide a general indication of resource conditions within their habitat.

METHODS

The Santa Monica Mountains comprise approximately 64,000 ha of chaparral-covered coastal mountains situated within the transverse ranges of southern California. Elevations range from sea level to 948 m. Steep rock outcrops of volcanic and sedimentary origin accentuate the crest of the mountains. These rock outcrops are the preferred nesting sites of golden eagles.

The first golden eagle nest surveys were conducted on foot, requiring long climbs through thick chaparral to reach vantage points from which nest sites could be observed. Some nests were too far from observation points to determine nest status, while others were simply inaccessible due to rough terrain or private land ownership. One nest site eluded detection for years.

The difficulty in executing successful golden eagle nest surveys on foot led to the use of helicopters in monitoring population and nest status. The helicopter provided a systematic observation of all known and potential nest sites during each annual survey. Initially, aerial photos were studied to define areas that had cliff faces which met criteria for potential nest sites. The criteria included vertical cliffs of sedimentary or volcanic origin in remote areas, with eroded "potholes" high on the rock, providing relative safety for the young. Once potential nesting areas were identified, a flight plan which specified the survey route was developed to maximize the efficiency of flight time. Limited funds made it necessary to conduct flights when there was a high probability that young would be present. Initial surveys were made at various times between January and June, until an optimal period for conducting the survey was determined. Flight dates were standardized for consistency of data collection between 15 April and 1 May, when young were usually well developed and easy to see in the nest. The earliest a nestling golden eagle had been observed in a nest was 30 January, and the latest 30 June. Nesting chronology in other areas may differ significantly.

The hazardous nature of helicopter nest site surveys requires the incorporation of certain precautions and techniques as an integral part of operational procedures to ensure safety. All personnel participating in the survey need to be trained in working safely in and around helicopters. Pre-flight briefings for the helicopter pilot, including the intended route of the flight displayed on a map, and techniques for approaching nest sites to minimize disturbance to adult birds and nestlings, should be stressed.

Small helicopters, including the Hughes 500 and Bell 206, were used in the surveys. The flight characteristics of these aircraft are well suited for the tight, slow flying required for direct views of the nests. Larger helicopters generate rotor winds that can blow nestlings from their nests or adults into the rocks below. Even with the small helicopters used in the survey, care was taken when approaching a potential nest site. Approaches were made slowly and directly at the cliff, at the same elevation as the nest. This allowed the birds time to see the helicopter and react without damaging eggs, nestlings, or the aircraft. During our survey, adult golden eagles would often abandon the nest site as the helicopter approached, revealing the nest location. As the helicopter hovered next to the nest site for observation, nestlings would lay motionless in the nest as the adults soared nearby or perched on a nearby cliff.

Active golden eagle nest sites were located during the flight by scanning cliffs for "whitewash" excrement markings. Perch locations around the nest site were sometimes marked by "whitewash," indicating that a nest was nearby.

The ability of the helicopter to fly slowly, close to cliff faces, hover and turn within a short radius proved to be ideal for surveying the hundreds of "potholes" that dotted the cliff faces, all of which were potential golden eagle nest sites. While helicopter costs are higher than those for fixed-wing aircraft, their flight charcteristics provided optimal data collection. Hickman (1972) used fixed-wing aircraft to survey golden eagle nests in Idaho; however, in our case, the use of helicopters enabled us to obtain air speeds low enough for extended nest observations. This was a primary factor in ensuring the success of our data collection efforts at Santa Monica NRA.

RESULTS

Our use of helicopters in conducting the annual golden eagle nest site surveys increased the accuracy and consistency of data collection in comparison to previous ground surveys. New golden eagle nest sites were detected on Boney Mountain and a nest site was confirmed after four years of searching in the Palo Comado Canyon area (Fig. 1).

Since the survey's beginning in 1981, the number of golden eagle nest sites have generally remained constant, with the exception of the Malibu Canyon nest site which was abandoned in 1985. The annual survey included four golden eagle nest sites; Boney Mountain, Lobo Canyon, Malibu Canyon, and Palo Comado Canyon (designated as the Cheeseboro golden eagles in data summaries) (Fig. 1).

Dixon (1937) noted in his 30-year study of golden eagles in San Diego County that the eagles maintained territories of 36 square miles between nest sites with little or no overlap. Using the same territorial criteria of 36 square miles (57.9 km) for each pair of golden eagles in the Santa Monica Mountains, all suitable habitat is being utilized with very little

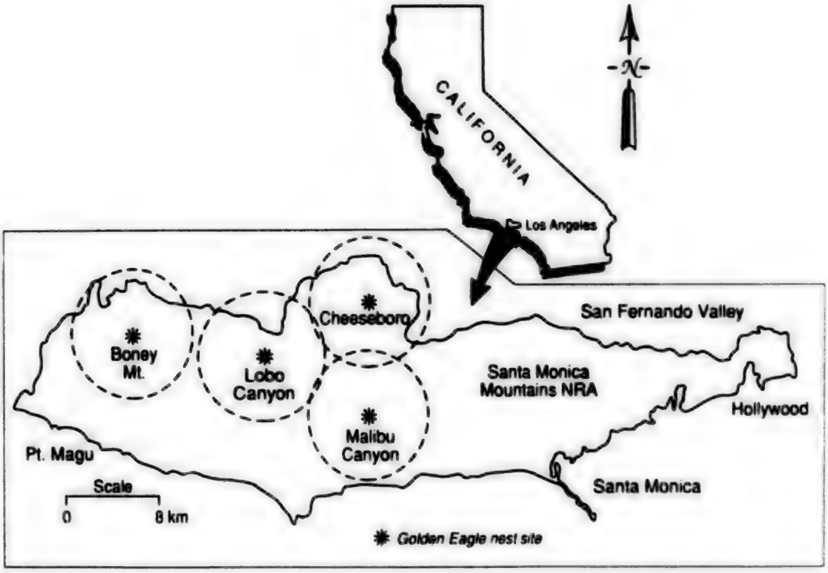

FIGURE 1. Golden eagle nest sites in Santa Monica Mountains National Recreation Area. Dotted lines include 36 square miles (57.9 km).

overlap (Fig. 1). Areas in the Santa Monica Mountains that do not have golden eagle nests lack rock formations for cliff nests, or suitable habitat has been disturbed by extensive urban development.

Although the number of nest sites were generally consistent, production at individual nests has varied significantly from year to year (Table 1). Data indicate a general decline in the number of golden eagle young produced per nest between 1981 and 1989. Nest site production ratios (number of young produced per nest per year) decreased from a high of 1.2 in 1982/1983 to a low of 0.33 in 1988/1989. Nest sites with the number of young produced noted as "unknown" in Table 1 were not used in the calculation of productivity ratios. The annual average productivity ratio for golden eagle nests in the Santa Monica Mountains is 0.625 young per nest. Comparable studies (Dixon 1937, Boeker and Ray 1971) found golden eagle productivity ratios ranging from 1.3 to 1.75 young produced per nest per year.

TABLE 1. Summary of golden eagle nest site surveys conducted at Santa Monica Mountains NRA, 1980-1989.

Year	Nest Site	Nest Occupied	Nest Active	No. of young
1980	Lobo Cyn.	yes	yes	1
1981	Boney Mtn.	yes	yes	1
1981	Cheeseboro	(Adults present/nest undetected)		
1981	Lobo Cyn.	yes	yes	1
1981	Malibu Cyn.	yes	(unknown)	
1982	Boney Mtn.	yes	yes	2
1982	Cheeseboro	(Adults present/nest undetected)		
1982	Lobo Cyn.	yes	yes	2
1982	Malibu Cyn.	yes	(unknown)	
1983	Boney Mtn.	yes	no	nest failure
1983	Cheeseboro	(Adults present/nest undetected)		
1983	Lobo Cyn.	yes	yes	1
1983	Malibu Cyn.	yes	yes	1
1984	Boney Mtn.	yes	no	nest failure
1984	Cheeseboro	(Adults present/nest undetected)		
1984	Lobo Cyn.	yes	yes	1
1984	Malibu Cyn.	yes	yes	1
1985	Boney Mtn.	yes	no	nest failure
1985	Cheeseboro	yes	no	nest failure
1985	Lobo Cyn.	yes	yes	1
1985	Malibu Cyn.	no	no	nest abandoned
1986	Boney Mtn.	yes	no	nest failure
1986	Cheeseboro	yes	yes	1
1986	Lobo Cyn.	yes	yes	1
1986	Malibu Cyn.	no	no	nest abandoned
1987	Malibu Cyn.	no	no	nest abandoned
1987	Boney Mtn.	yes	no	nest failure
1987	Cheeseboro	(Adults present/nest undetected)		
1987	Lobo Cyn.	yes	no	nest failure
1988	Boney Mtn.	yes	yes	1
1988	Cheeseboro	yes	(unknown)	
1988	Lobo Cyn.	yes	yes	1
1988	Malibu Cyn.	(1 adult present at old nest site)		
1989	Boney Mtn.	yes	no	nest failure
1989	Cheeseboro	yes	no	nest failure
1989	Lobo Cyn.	yes	no	nest failure
1989	Malibu Cyn.	no	no	

DISCUSSION

Golden eagles have, in general, maintained a consistent population in the Santa Moncia Mountains through the nine years of this study. The only exception was the Malibu Creek nest site which was abandoned in 1985. The loss of this nesting pair represented a 25% decrease in the overall population in the Santa Monica Mountains. This nest had been active since at least 1970 (Knight, unpublished data). In 1988, a single adult was observed flying from the old nest site, perhaps trying to re-establish the nest. However, the 1989 survey indicated that the nest was not re-established.

Annual production rates of the nest sites have declined steadily since the beginning of the study. The annual productivity ratio of 0.625 young per nest calculated for the nine years of the study is significantly lower than productivity ratios calculated in other golden eagle studies (Dixon, 1937; Boeker and Ray, 1971). Total nest failures have become more common in recent surveys conducted in the Santa Monica Mountains. Drought conditions in the mountains have persisted since 1987 and may be affecting prey populations. In the 1989 survey, all nests were empty with adults present only on the Boney Mountain and Lobo Canyon nest sites. Extreme temperatures (over 100° F) for extended periods in the early part of the nesting season may have contributed to complete nest site mortality.

The most obvious impact to the golden eagles over the period of the study has been human intrusion into their habitat. Large-scale urbanization has destroyed thousands of hectares of prime grassland hunting habitat near nest sites. Small lot developments and individual residences, often constructed less than half a kilometer from nest sites, increased the level of human intrusion on nest sites. Santa Monica NRA has become more popular with urban residents, with increasing numbers of people hiking and climbing on rocks near nest sites.

CONCLUSIONS

All of the previously discussed impacts could serve as explanations for the decline of the golden eagle population in the Santa Monica Mountains, or all of these factors may be working in combination to create an environment unsuitable for the species. Whatever the cause, if the remaining population is to be maintained, measures must be taken by the National Park Service and local government planning agencies to protect the nest sites from further intrusion by development. The park must ensure that trails are routed away from nest sites, and that visitors are kept from the vicinity of nests during the nesting season.

It is recommended that the helicopter nest site survey continue on an annual basis to monitor the status of the golden eagle population in the

Santa Monica Mountains. Additional studies should include banding and radio telemetry of young eagles to determine survival rates and dispersal characteristics.

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Breeding Distribution of the Marbled Murrelet in Redwood National Park and Vicinity During 1988

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Abstract. We report on an intensive research effort to determine the present status of the marbled murrelet (*Brachyramphus marmoratus*) in the vicinity of Redwood National Park, in Del Norte and northern Humboldt Counties, California. This seabird is primarily an inhabitant of the nearshore waters and adjacent coastal redwood forests of the northern half of the state. Little is known of its ecology on land. Using remote sensing techniques, we mapped all old and mature forests, the potential habitat for the species. We then conducted systematic surveys of stands selected from the above inventory, recording the number of detections of birds, their behavior, and various vegetative aspects of the stands. Within the area including Redwood National Park, the nearby redwood state parks and private lands, 120 morning counts were conducted on 54 transects, with murrelets detected on 70% (38) of the transects. Bird distribution in the study area was patchy and essentially restricted to old-growth redwood forests.

INTRODUCTION

One of the last and most enduring mysteries of ornithology is the life history of the marbled murrelet (*Brachyramphus marmoratus*). Although fairly common in nearshore marine waters, the murrelet has kept its breeding grounds amazingly secret. The species occurs around the Pacific rim from Korea and Japan, through Alaska, and southward into central California. It is considered to be a marine species, although it is known to use inland lakes year-round in the Pacific Northwest (Carter and Sealy, 1986). On the west coast of North America, in the southern part of its range, it is thought to nest only in trees in old-growth forests (Sowls et al., 1980; Sealy and Carter, 1984), although through 1988 only four tree nests have been found anywhere in the species' range. In California, all evidence points to the bird nesting primarily in old-growth redwood (*Sequoia sempervirens*) forests, as summarized by Carter and Erickson (1988). These forests have been much reduced over the past 200 years, with old-growth stands now making up only about 10% of the former range of the redwood (L. Fox, pers. comm.). These remnants are still being harvested. Thus the status and continued persistence of the California murrelet population may be tied to these remnant forests. Despite this possibility, there have been no systematic inland surveys describing their breeding distribution or habitat use in the state. Research conducted on the species has focused primarily on observations at sea, including distribution (Sowls et al., 1980; Sealy and Carter, 1984), inferred breeding biology (Sealy 1975a; Hirsch et al., 1981), and feeding ecology (Sealy, 1975b). Nest and egg descriptions have included records from northern latitudes, where the species nests on the ground on treeless islands (Simons, 1980; Johnston and Carter, 1985), and from the southern parts of the range, where nests in trees have been found (Kyzyakin, 1963; Binford et al., 1975). Historical information on this species in California is summarized by Carter and Erickson (1988).

The purpose of this paper is to document the species' occurrence in Redwood National Park and surrounding areas, as a portion of a census that included stands from the Santa Cruz Mountains, Santa Cruz County, to the Oregon border (Paton and Ralph, 1988).

METHODS

Stand Selection

Although marbled murrelets have been reported at sites as far inland as 75 km in Washington (Carter and Sealy, 1986), the farthest inland murrelet site in California appears to be 39 km inland at Grizzly Creek Redwoods State Park. Therefore, we surveyed sites up to 40 km inland. Data collected by us at Redwood Experimental Forest near Klamath, California (Paton et al., in review) suggested that murrelets appear to be closely associated with closed canopy old-growth redwood stands. We therefore weighted our sampling scheme towards that habitat type, although we also surveyed younger stands.

In order to identify potential sites, we used data from a distributional study of the coastal redwood forests based on remote sensing techniques (L. Fox, pers. comm.). Maps were based on aerial photos taken at 19,800 m with high-definition Aerochrome infrared SO-131 film by a RC10 sensor with a six-inch focal length. Habitat types delineated by the photos include: (1) old-growth redwood, > 70% canopy closure; (2) old-growth redwood, < 70% canopy closure; (3) old-growth redwood/Douglas-fir (*Pseudotsuga menziesii*); (4) mature redwood; and (5) young or clearcut redwoods. Fox (pers. comm.) defined old-growth as stands in which some trees pre-date the arrival of Europeans in the 1700s.

Bird Surveys

On the basis of the distribution of vegetation, we laid out 54 transects (Fig. 1). Surveys, following a modified protocol of Paton et al. (1988), were conducted from 15 May to 15 August 1988 in order to include the seasonal peak of murrelet activity in the stand. Each transect was made up of 8-13 fixed stations surveyed in one morning (Table 1). Depending upon road or trail conditions, stations were placed 250-1000 m apart, with 250 m spacing between stations along trails, 500 m along rough roads, and usually 1 km along paved roads. In order to encompass the peak of daily bird activity, counts began 45 minutes before sunrise and continued for one hour and 30 minutes after sunrise. Each station was surveyed for 10 minutes, and each transect surveyed at least twice during the study (see Table 1). The order of stations was reversed each time the transect was done, so as to have each station surveyed at different times during the morning.

The basic datum recorded was the "detection," defined as the sighting or hearing of one or more birds flying together. Most birds were detected as they rapidly flew overhead, usually within about 200 m of the ground. Morning surveys were conducted 120 times over the 54 transects (Table 1). Birds were detected on 38 (70%) of the transects.

We do not yet know the relation between the number of detections and the actual number of birds using a particular stand. However, we believe the detections are an index of bird abundance. Due to variations in visibility at different sites, we believe the detections, rather than the number of birds seen or heard, are a more reliable estimator of the difference in relative abundance among transect locations. A stand with one detection compared to another stand with 100 detections during the same time period is assumed to have fewer birds. The differences in relative abundance among stands are preliminary results based on only a few visits to each site. An observer will miss birds at stands where murrelets are present, and some transects with only a few detections could have had many more birds.

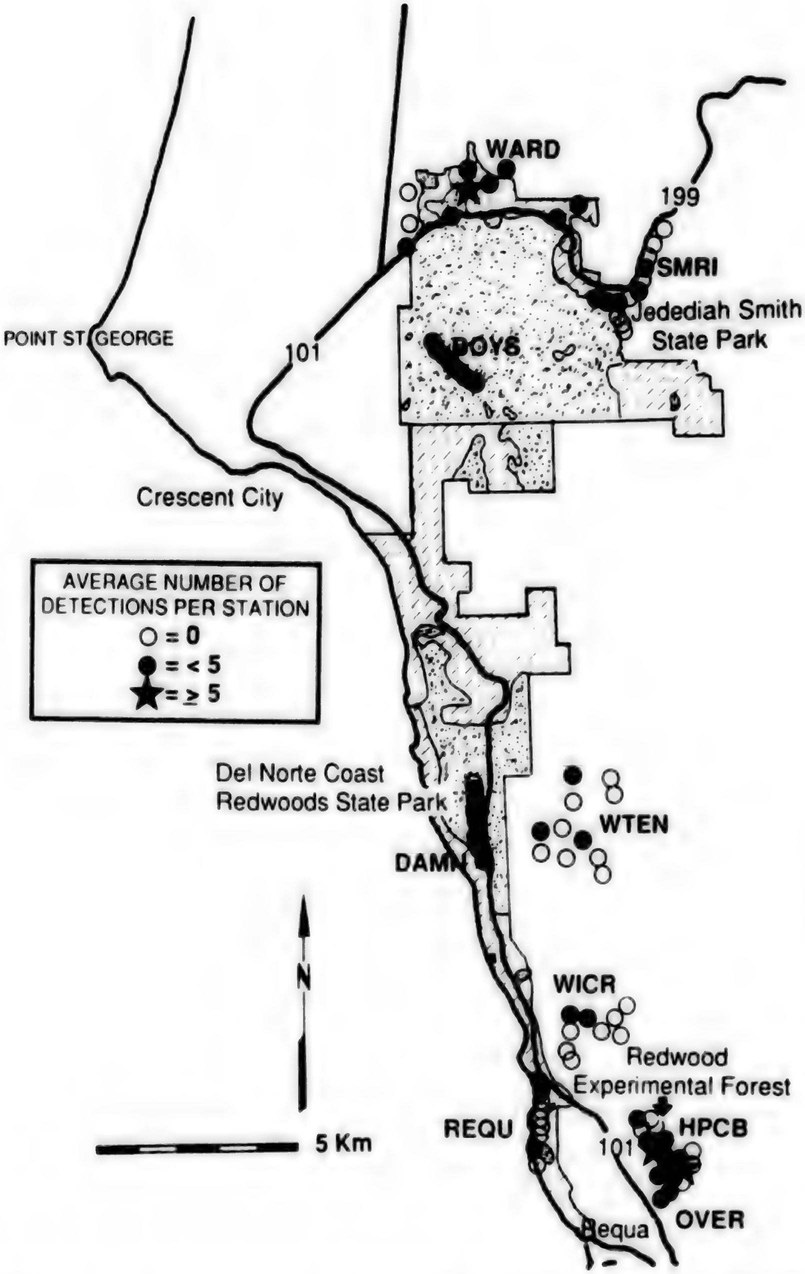

FIGURE 1. Map of study sites at Jedediah Smith, Del Norte Coast, Wilson Creek and Redwood Experimental Forest areas, showing each station with an indication of the number of birds detected. The mnemonic for the transect name is indicated (see Table 1); park extent is shown by cross hatching, and old-growth forests by stipling.

Table 1. Location and number of marbled murrelets detected during the 1988 survey. The Code indicates the name on Figure 1, the Type refers to: CARL = car long, 1 km between stations; CARS = car short, 0.5 km between stations; WALK = walking transect, 250 m between stations; and BIKE = 0.5 km between points on a bike. Also shown is the number of stations done at each transect (Sta), the number of detections (Det), the detections per station, and the number of days that the transect was censused. The transects are arranged from the most northern to the most southern in the study area.

Transect	Code	County	Owner	Type	Sta	Det	Det /Sta	No. of days
Rowdy Creek	ROCR	Del Norte	Private-Many	CARL	33	2	0.0600	3
Kermit Miller Exch.	KEMI	Del Norte	US Forest Ser.	CARS	41	0	0.0000	4
Smith River/Hiouchi	SMRI	Del Norte	Public/Private	CARL	22	7	0.3150	2
Walker Road	WARD	Del Norte	State Park	CARL	20	29	1.4500	2
Myrtle Creek	MYCR	Del Norte	US Forest Ser.	WALK	30	11	0.3667	3
Boy Scout Trail	BOYS	Del Norte	State Park	WALK	22	55	2.5000	2
Damnation Tr. (Bluff)	DAMN	Del Norte	Nat'l Park Ser.	BIKE	21	24	1.1428	2
High Prairie-Yurok	HPCB	Del Norte	US Forest Ser.	CARS	55	134	2.4340	5
Overlook (Yurok E.)	OVER	Del Norte	US Forest Ser.	CARS	44	71	1.6150	4
Requa (N. Klam. Riv.)	REQU	Del Norte	Nat'l Park Ser.	WALK	22	3	0.1350	2
S-A Forestry Headqrt.	FOHE	Del Norte	Simpson	CARL	18	1	0.0550	2
S-A H500/P500	HFPF	Del Norte	Simpson	CARL	19	0	0.0000	2
S-A K-One	KONE	Del Norte	Simpson	CARL	20	0	0.0000	2
S-A U-Ten	UTEN	Del Norte	Simpson	CARL	18	0	0.0000	2
S-A W-Ten	WTEN	Del Norte	Simpson	CARL	20	11	0.5500	2
S-A Upper Corners	UPCO	Del Norte	Simpson	CARL	20	0	0.0000	2
S-A Wilson Creek	WICR	Del Norte	Simpson	CARL	16	14	0.8750	2
Alder Camp (N. Klam.)	ALDR	Del Norte	Nat'l Park Ser.	CARL	22	10	0.4550	2
Camp Klamath	CAKL	Del Norte	Private-Many	CARL	22	2	0.0900	2
Klamath	KLAM	Del Norte	Nat'l Park Ser.	CARL	22	5	0.2250	2
Red Mountain	REMO	Del Norte	US Forest Ser.	CARL	22	0	0.0000	2

Table 1. (continued)

Transect	Code	County	Owner	Type	Sta	Det	Det /Sta	No. of days
S-A B-900	BNHU	Humboldt	Simpson	CARL	22	0	0.0000	2
North West Ridge	NWRI	Humboldt	State Park	WALK	20	43	2.1500	2
James Irvine	JITR	Humboldt	State Park	WALK	17	148	8.7059	2
North By-Pass	BYPN	Humboldt	Nat'l Park Ser.	CARL	22	0	0.0000	2
South By-Pass	BYPS	Humboldt	Nat'l Park Ser.	CARL	20	0	0.0000	2
West Ridge Trail	WERI	Humboldt	State Park	WALK	19	67	3.5263	2
Gold Bluff Road	GBBR	Humboldt	State Park	WALK	20	34	1.7000	2
Hope Creek-Ten Tappo	HOPE	Humboldt	State Park	WALK	20	80	4.0000	2
Prairie Creek Hwy 101	PHWY	Humboldt	State Park	CARL	20	109	5.4500	2
A-9 Road	ANIR	Humboldt	Nat'l Park Ser.	CARS	27	14	0.5167	3
East C-line Road	ECLR	Humboldt	Nat'l Park Ser.	CARS	33	8	0.2400	3
Lady Bird Johnson	LBJG	Humboldt	Nat'l Park Ser.	WALK	27	16	0.5933	3
South Operations	SOCT	Humboldt	Nat'l Park Ser.	WALK	33	11	0.3300	3
West Side Access Rd.	WSAR	Humboldt	Nat'l Park Ser.	CARS	33	1	0.0300	3
Cal-Barrel Rd.	CABA	Humboldt	State Park	CARL	17	39	2.2941	2
Lost Man Picnic Area	LLMC	Humboldt	Nat'l Park Ser.	CARS	20	69	3.4500	2
Lower Redwood Creek	LRCK	Humboldt	Nat'l Park Ser.	WALK	33	218	6.6060	3
Skunk Cabbage Creek	SKNK	Humboldt	Nat'l Park Ser.	WALK	22	44	1.9950	2
Tall Trees Grove	TTGR	Humboldt	Nat'l Park Ser.	WALK	21	41	1.9524	2
Geneva Road	GNVA	Humboldt	Nat'l Park Ser.	CARS	22	5	0.2250	2
Bald Hills Road	BALD	Humboldt	Nat'l Park Ser.	CARL	22	28	1.2700	2
Orick Horse Trail	HORS	Humboldt	Nat'l Park Ser.	WALK	22	23	1.0450	2
Stone Lagoon	STLA	Humboldt	Private-Many	CARL	22	0	0.0000	2
Lagoons (Stone-Big)	LAGS	Humboldt	State Park	CARS	23	1	0.0435	2
Patrick's Point	PAPT	Humboldt	State Park	BIKE	18	1	0.0550	2

Table 1. (continued)

Transect	Code	County	Owner	Type	Sta	Det	Det /Sta	No. of days
Devil's Creek	DECR	Humboldt	Nat'l Park Ser.	WALK	22	12	0.5450	2
M-Line (S. RNP)	RNPM	Humboldt	Nat'l Park Ser.	CARL	22	0	0.0000	2
LP #5 R-Line	LPRL	Humboldt	Louisiana-Pac.	CARL	22	0	0.0000	2
LP #2 Little River 2	LRII	Humboldt	Louisiana-Pac.	CARL	20	0	0.0000	2
LP #3 M-Line	LPML	Humboldt	Louisiana-Pac.	CARL	22	5	0.2250	2
LP #6 Maple Creek	LPMC	Humboldt	Louisiana-Pac.	CARL	21	0	0.0000	2
LP #4 T-Line	LPTL	Humboldt	Louisiana-Pac.	CARL	22	0	0.0000	2
LP #1 A-Line	ALIN	Humboldt	Louisiana-Pac.	CARL	22	0	0.0000	2

RESULTS

In our survey of the state, we identified three primary areas of murrelet breeding activity: (1) the Crescent City area south to Redwood Creek in Redwood National Park; (2) Pacific Lumber Company lands east of southern Humboldt Bay to Humboldt Redwoods State Park on the Eel River; and (3) state parks in southern San Mateo and northern Santa Cruz Counties. In this paper, we will discuss the survey results in Del Norte and northern Humboldt Counties.

Jedediah Smith State Park and Vicinity

The transects in northern Del Norte County had a moderate to high detection rate, concentrated in the immediate area of the Smith River (Fig. 1). Within the state park itself, the Walker Road (WARD) transect had the most detections, averaging 1.5 detections per station. The Boy Scout Trail transect (BOYS), which enters the center of the old-growth in the park, had birds detected at all but one station, suggesting relatively high use of the stand center.

One of the most interesting observations involved murrelets detected in the Myrtle Creek drainage (MYCR) of Six Rivers National Forest, where birds had not been previously suspected. This is the only transect in the entire state where we detected birds in an area not dominated by redwood trees. The larger trees were confined near the creek, in a drainage that consisted primarily of Douglas-fir and Port Orford cedar (*Chamaecyparis lawsoniana*). No birds were detected farther up the Main Fork of the Smith River than Myrtle Creek, and no transects were located on the South Fork of the Smith. The only marked directionality to flights observed was at Myrtle Creek, where birds generally headed south, probably towards the mouth of the Smith River.

We did not detect birds at some places where murrelets have recently been heard, such as Hutsinpillar Creek, at the north end of the Rowdy Creek transect (ROCR) (Carter and Erickson, 1988), and at Camp Lincoln on the Kings Valley Road (Dan Scott, pers. comm.).

Del Norte Coast Redwoods State Park and Wilson Creek Area

This state park is a relatively narrow coastal strip of old-growth redwood. We had modest numbers of murrelet detections at most stations on the Damnation Trail transect (DAMN), at the southern end of the park west of Highway 101. A few birds were also detected on the WTEN transect in Simpson Timber Company lands, just to the east of Del Norte Redwoods. The low number of detections on the WTEN transect suggests that many of the birds detected on the coastal transects are probably nesting in the state park, and not proceeding farther east. The Simpson Timber Company lands to the east of the park are heavily fragmented, and only small patches of old-growth redwood remain. The Wilson Creek

(WICR) transect on Simpson land included some old-growth islands, and two of the stations near Wilson Creek had a total of 14 detections.

Redwood Experimental Forest to the Mouth of the Klamath River

The Requa transect (REQU), west of the Yurok Loop Trail in Redwood National Park, had only a few detections, and those could have come from the Redwood Experimental Forest (HPCB and OVER transects). We detected many murrelets in tracts owned by the U.S. Forest Service, with an average of 2.4 detections per station on the High Prairie/Yurok transect (HPCB). Birds here and at the nearby Overlook transect (OVER) were generally detected as they flew down High Prairie Creek in a southwest direction towards the ocean. It appeared that most birds were confined to the Research Natural Area on the Experimental Forest. Several all-morning stationary counts at a single station were conducted here in late July, with a peak of 146 detections at a station along the flight corridor to the Natural Area.

Of the six other transects on the highly-fragmented Simpson lands in Del Norte and Humboldt Counties, only one had murrelets besides the WTEN and WICR transects. The S-A Forestry Headquarters (FOHE) transect had one detection on 29 July, but generally the Simpson lands appear to lack substantial numbers in the areas we were able to survey. We recorded two detections west of Klamath and murrelets had been detected flying down Terwer Valley on 10 July 1987 (Carter and Erickson 1988). However, we have no data to indicate the breeding areas of those birds.

The Alder Camp transect (ALDR) in Redwood National Park was somewhat active with birds either originating from the adjacent old-growth redwood stands or flying from farther inland. The lack of detections along the Camp Klamath (CAKL) transect suggests that the point of origin of the birds on the ALDR Transect was probably the redwoods just east of Flint Rock Head.

Prairie Creek State Park

Many birds were found here, with the James Irvine Trail transect (JITR) having an average of 8.7 detections per station, the highest of any transect in the state (Fig. 2). Within a few kilometers, the Hope Creek-Ten Tappo (HOPE), Cal-Barrel Road (CABA), West Ridge Trail (WERI), North West Ridge (NWRI), and Prairie Creek Highway 101 (PHWY) transects all ranked in the top 15 transects statewide, based on the relative number of murrelet detections. Murrelets in these transects usually flew northwest, out of drainages and towards the sea, with birds apparently funneling out of the Fern Canyon area down Godwood Creek.

In contrast, just to the east of the park, the North By-Pass (BYPN) and South By-Pass (BYPS) transects had no detections. This was quite unexpected, as the two transects were adjacent to the old-growth forests con-

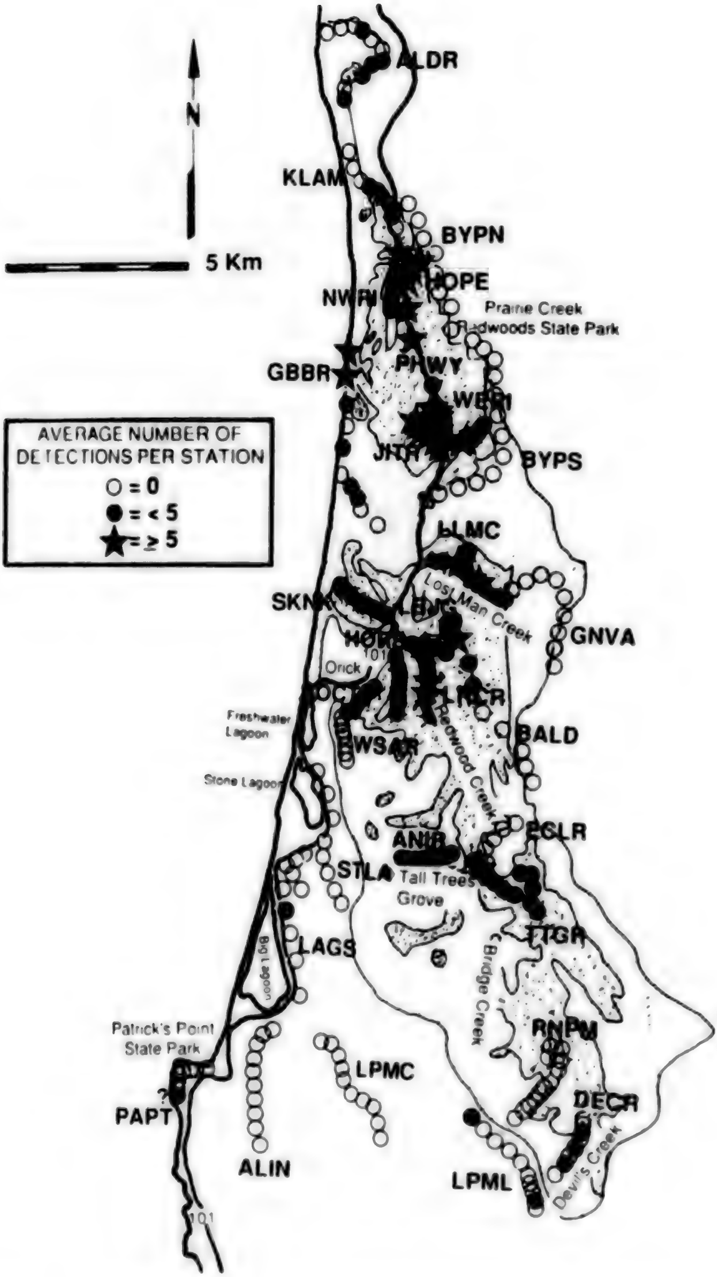

FIGURE 2. Map of study sites at Prairie Creek and Redwood National Park areas, showing each station with an indication of the number of birds detected. The mnemonic for the transect name is indicated (see Table 1); park extent is shown by cross hatching, and old-growth forests by stipling.

taining the transects mentioned above. To the east of the park, in the area of the two By-Pass transects, virtually no areas of old-growth forest remain because it has been thoroughly harvested, primarily in the past 20 years.

Redwood National Park

Just to the south of the Prairie Creek area, the Lost Man Creek drainage (LLMC) had many murrelet detections, with the highest numbers at the confluence of the two main forks of the creek (Fig. 2). Flight paths in this area tended to be along drainages, either northeast or east. Interestingly, no birds were detected along the upper portion of the nearby Geneva Road (GNVA) transect once the transect left the old-growth redwoods, and went into a second growth, hardwood-dominated forest. With the detections on the LLMC and GNVA transects confined to the old-growth redwood areas along the drainages, it appeared that the birds were not nesting to the east.

The Lady Bird Johnson Grove (LBJG) was an area with many murrelet detections, but the adjacent Lower Redwood Creek transect (LRCR) had the most detections of any transect in the park, averaging 6.6 detections per station (the second highest number in the state). This is probably the result of relatively large numbers of murrelets nesting in this area and using the drainage as a flight corridor. The Bald Hills Road (BALD) transect had detections at stations in old-growth redwood, but none on the ridge where the Bald Hills Road moves into second-growth forests. The Horse Trail (HORS) transect had detections at most stations. We speculate that either birds were nesting nearby or flying over the ridge from Lower Redwood Creek. In all likelihood both were involved, since birds were observed to be flying in all directions on this transect. The Tall Trees Grove (TTGR) had relatively high numbers of detections, 2.0 per station, with most birds confined to stations near Redwood Creek. Although there were many detections at TTGR and the A-9 Road (ANIR), we had almost no detections on the transects closer to the coast: Stone Lagoon (STLA), Lagoons (LAGS), and West Side Access Road (WSAR).

Area east of Trinidad

Murrelet detections were relatively low in this area, with only one apparent concentration of birds in the Devil's Creek (DECR) drainage in Redwood National Park. Birds from DECR could be responsible for the few birds detected on the Louisiana-Pacific (LP) Company's M-Line (LPML) transect. All the transects on LP Company lands in this area (LPMC, LPRL, LPML, LPTL, ALIN, LRII) had few detections. Transects on these private lands all passed through second growth habitat, while the only large stand of old-growth in the area was on the DECR transect in the park.

DISCUSSION AND CONCLUSIONS

There are limits to the type of survey data that can be collected on such an elusive bird as the marbled murrelet. While we believe large numbers of detections indicate a fairly high murrelet population, we do not suggest that there is a 1:1 relationship between the number of detections and the number of birds in an area. In all probability, we failed to detect murrelets on some transects where the birds occurred, especially when the area may have supported only a few pairs. Little is known about the detectability of this species when comparing an isolated pair to a large concentration. There is a possibility that isolated pairs might be quiet and secretive, while large groups are vocal and social.

Based on preliminary surveys conducted in Oregon (Kim Nelson, pers. comm.) and Washington (Eric Cummings, pers. comm.), the California populations in the vicinity of Redwood National Park are probably the largest south of the Puget Sound area. If the murrelet populations to the north continue to decline, and the little remaining old-growth is removed in coastal Oregon and Washington, the California population will become increasingly isolated. We have no information about the mobility of the species, but there should be concern about possible genetic isolation.

The value of the population now preserved within the boundaries of Redwood National Park, and nearby portions of the California State Park System, is very high. Clearly, the park is one of the centers of murrelet abundance in the state. The lack of detections along the coast on STLA, LAGS, and WSAR transects, and the abundance of birds on Lower Redwood Creek just to the east, suggests that they might be flying north following the Redwood Creek drainage from TTGR rather than heading due west directly towards the ocean. The lack of detections to the east of Redwood National Park and Prairie Creek State Park, with many detections in the parks themselves, is highly suggestive that murrelet nesting activity is confined to the park, primarily in a closed canopy, old-growth redwood forest. Other than small populations in private timber company lands to the southeast, these are the only murrelets remaining in the state, except for an even smaller, isolated concentration in the Santa Cruz Mountains, 200 km to the south.

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A Comparative Analysis of Mammal Inventory Data for California's National Parks

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Abstract. Increasing attention is being given to the role of the national parks in conserving biological diversity. Central to understanding this role is the availability of current, standardized species inventory data. This paper discusses basic problems concerning the scope and accessibility of park mammal occurrence data. Mammal species lists, derived from park records by other researchers, are compared for consistency with park species lists provided by 16 National Park Service (NPS) units in California. Results of these analyses indicate numerous discrepancies in agreement concerning park mammal species composition. All apparent extinctions are disavowed as mistaken conclusions arising in part from inconsistent record-keeping by many parks. Alternative species lists, all supposedly drawn from park files, disagree in composition. These discrepancies are most likely the result of great differences in the scope and quality of species inventory data among parks, as well as the accessibility of much of this information. We recommend that a centralized system for maintaining species inventory data be established to facilitate a standardized approach to record-keeping and to provide information essential to analyzing the present conservation value of parks.

INTRODUCTION

Of growing appreciation to resource managers, scientists, and conservationists is the need to conserve biological diversity in fully protected reserves (May, 1975; Soulé and Wilcox, 1980; Wilson, 1988). Although the national parks have not been traditionally designed or operated with biological diversity as a primary goal, increasing attention is now being given to assessing their role in preserving national biodiversity (Bureau of International Organization Affairs, 1984; Office of Technology Assessment, 1987; Shen, 1987; Scott et al., 1987). Central to understanding this role is the availability of current, accurate, and standardized species inventory data.

Species inventory data are critical not only for the management of biological diversity; they also allow researchers to better address pressing questions concerning the design and location of future reserves. While the national parks serve as natural laboratories for assessing different approaches to conservation, the number of plants and animals occurring has not yet been accurately determined. The immediate threats to wildlife and the need to determine current levels of protection call for the availability of easily accessible, standardized species inventory databases. The use of comparable species inventories will help us determine the number of species currently under full protection in parks, how vulnerable these species are to extinction, and how well these reserves function to prevent extinctions.

Many parks maintain collections and sighting records, as well as species lists from a variety of sources. However, systematic inventory activities have historically received low funding priority. Many parks have not been surveyed since the early 1930s, when George Wright initiated the first preliminary wildlife surveys in the national parks. Much of the data available today on park mammal composition are records dating back to this time. Many of the contemporary accounts are nonempirical; a number of lists are based in part or in whole upon the distributional accounts of Ingles (1965) and Grinnell and Storer (1924). Recent records of occurrence are often based on visitor sightings or irregular reports of species by park personnel. Many of the small, fossorial, or nocturnal species, and often the more common ones, may go unreported.

We identify two main problems which the current state of mammal occurrence data presents for comparative research: (1) due to the inconsistent nature of reporting species occurrence information, these data are not comparable among parks; and (2) in many parks, natural history data, including species sighting records, are not kept in a centralized location; they are often scattered among various offices, files, and documents. These data are, therefore, not readily accessible to managers and researchers. Some parks have multiple species lists that were designed for different uses at different times. It is often difficult to determine how the lists were assembled, which taxonomic references were used, or how current and reliable they are.

Confusion and misinterpretation of important biological processes and patterns central to conservation efforts can arise when the available occurrence data are treated as comparable among parks. A case in point is a highly publicized report of mammalian extinctions in western North American national parks (Newmark, 1987). Newmark (1986) derived species lists for 24 national parks, based upon park sighting records, conversations with select park officials, and published accounts of species occurrences. He then analyzed these records for recent (post-establishment) extinctions of carnivore, artiodactyl, and lagomorph species, assuming that a species had gone extinct from a park if there were no recorded sightings within the 10 years preceding 1984. From this analysis, Newmark (1987) inferred 39 cases of natural, post-establishment extinctions of large mammals in 14 western national parks. If true, this would indicate a failure of the parks to preserve viable populations of many mammal species.

However, when the actual status of each of these species was checked with certain NPS biologists, managers, and other experts, it was found that in 21 of these cases there had been confirmed sightings of the species within the parks in the last five to 10 years (Quinn et al., 1989). Another six species are reported as probably extant, and in seven cases there is little information to suggest that populations ever occurred in the park. In 34 of the 39 cases, populations are known from adjacent national forests. In all, Quinn et al. (1989) found three possible (but still questionable) cases of natural post-establishment extinctions in the 14 parks studied by Newmark.

To resolve the problems mentioned above, we have initiated a long-term project to collect and standardize mammal inventory data for the national parks, along with numerous other protected areas. In this paper, we illustrate the two main problems with the currently available mammal species occurrence data for the national parks in California. We analyze the apparent cases of extinctions found by Newmark (1987) for California mammals and discuss how such errors in his conclusion are possible, and compare for consistency alternate datasets of mammalian occurrence in several parks. We conclude with a summary of the species lists we have collected to date for 16 of the 18 NPS units in California, including recent revisions for several parks along with supplemental information provided by Park Service biologists.

The data presented herein do not necessarily reflect all natural history information that may be available in park files. Much of this information is still unavailable to researchers, and a considerable amount of work remains to be done.

We hope to accomplish two goals with this paper: (1) to stimulate an interest in reviewing and updating the current species lists; and (2) to call attention to the need for a unified approach and standard protocol for collecting and managing biological inventory information. Our purpose is not to criticize the hard work and ongoing efforts of biologists to establish useful species inventories, but to identify and discuss the discrepancies that we have encountered in attempting to review currently available mammal inventory lists.

METHODS

Mammal species lists were provided by biologists and managers from 16 of the 18 California National Park Service units. Since the beginning of this project in 1985, we have interviewed personnel familiar with the mammalian faunas of the parks, and have received current updates to these species lists from eight parks. Nomenclature was standardized to Jones et al. (1986), using the phylogenetic hierarchy of Jones et al. (1982). Each park mammal list was entered into a computerized database using dBASE III+. Hall (1981) was referenced to resolve disagreements between binomial classifications of species on the park lists with those of Jones et al. (1986). Species lists for all 16 parks are presented in Appendix 1.

Only terrestrial, native species were used in the present analysis, although all species (including exotic and feral animals as well as marine mammals) are listed in the appendix.

Since Newmark (1987) assessed extinctions based on the dates of the most recent recorded sightings he was able to find, evidence for more recent known incidences of occurrence for each case was researched by contacting a number of NPS and U.S. Forest Service biologists and managers (see Quinn et al. 1989). Newmark's (1986) park mammal lists contain all species for which the author was able to find any records of occurrence; it also identifies species supposedly extinct before park establishment and after.

Species occurrence data for individual parks were examined for consistency by comparing park lists with lists derived entirely or in part from park records by other researchers. Alternate mammal lists included those for Lava Beds National Monument and Lassen Volcanic National Park assembled by Newmark (1986) from park records and published accounts, and a study based upon park records by Van Gelder (1982). Van Gelder's study includes more of California's parks than Newmark's, but does not cover the shrews and moles, bats, and most rodent species. The dates of park lists and revisions were checked against the dates for these other lists. Nomenclature of all lists used for comparison were standardized to Jones et al. (1986), eliminating the problem of finding differences among lists due to taxonomic reasons. These dates of the park lists are compatible for comparison with those of these other researchers.

RESULTS

Comparability of Species Lists

The scope of the park mammal lists we received from the national parks in California varies greatly (Table 1). Some lists are reported to be relatively complete (e.g., Joshua Tree, Pinnacles, and Santa Monica); some include species for which there are no actual records of occurrence (these lists are probably based on range occurrence of species); some lists are apparently incomplete but only contain listings of verified sightings or specimens, while others reportedly contain only "representative" species. Only four of the lists we obtained were annotated. Very few included any supplemental information on how the lists were derived, what sources of information were used, which taxonomic references were used, or how current and reliable they are. Three of the lists gave only common names, though in most cases these were well known species and there was little doubt as to their correct identification (where doubt was present, the species was excluded from the database). Only one list contained subspecific identification.

TABLE 1. Mammal species lists obtained from 16 National Park Service units in California, along with park codes, source of the data, and comments, received by park personel, on quality and comprehensiveness of park mammal lists.

Park Name	Code	Area (Km^2)	Source	Comments
Cabrillo NM	CABR	0.58	unpublished list supplied by CABR	List is probably probably not complete.
Channel Islands NP	CHIS	506	unpublished park database, 1982	
Devils Postpile NM	DEPO	3.23	unpublished list compiled by J. L. Medeiros, 1981	Incomplete list.
Death Valley NM	DEVA	8368	unpublished list compiled by DEVA, 1988	All entries confirmed by published records.
Golden Gate NRA	GOGA	177	NPS, 1985	Provisional list.
Joshua Tree NM	JOTR	2270	unpublished list compiled by J.E. Cornely, 1985	Data from monument-wide surveys.

TABLE 1. (continued)

Park Name	Code	Area (Km^2)	Source	Comments
Lava Beds NM	LABE	187	unpublished list compiled by R. Houston, 1984	All inclusions confirmed.
Lassen Volcanic NP	LAVO	432	unpublished list compiled by LAVO	The list of mammals is not complete.
Muir Woods NM	MUWO	2.24	unpublished list compiled by MUWO	List includes only representative species
Pinnacles NM	PINN	65.6	Fellers and Arnold, 1987	Essentially complete.
Point Reyes NS	PORE	275	Fellers and Del 'Osso, 1985	Includes species not recorded in the park.
Redwood NP	REDW	426	unpublished list compiled by G. S. Lester, 1978	Includes species not recorded within the park.
Santa Monica NRA	SAMO	607	unpublished park database, 1988	Fairly complete.
Sequoia and Kings Canyon NP	SEKI	3430	Graber, in prep.	Data from actual records and reliable sources. Not complete.
Whiskeytown NRA	WHIS	172	NPS, 1975	
Yosemite NP	YOSE	3080	unpublished list compiled by YOSE	List of some of the more commonly seen mammals.

Not all lists were comparable taxonomically. Most were consistent with the classification of Ingles (1965), though a few more closely resembled Hall (1981). We chose to standardize our lists according to Jones et al. (1986) because it is a widely used classification among mammalogists and is revised periodically. This reference contains four additional terrestrial mammal species native to California than does Ingles (1965), and six more than Hall (1981); the binomial identification differs for 20 species at both the generic and species level from Ingles (1965). In most cases it was possible to resolve the taxonomic differences with Hall (1981), but this was a time-consuming process.

Record Keeping and Species Extinctions

We compared species lists from Lassen Volcanic National Park (NP) and Lava Beds National Monument (NM) with lists derived by Newmark (1986) for extant mammals in these parks (Table 2). Table 2 also identifies species listed by the parks that were determined to be extinct by Newmark (1987).

For Lassen Volcanic NP, Newmark (1987) cites seven species as extinct (all are on Lassen's species list), six of which have supposedly disappeared since the park's establishment in 1907. These include the black-tailed jackrabbit, Nuttall's cottontail, beaver, gray fox, ringtail, ermine, and spotted skunk. Newmark reports the most recent sighting records for these species as ranging from 1957 to 1974. Though the park's altitudinal range

TABLE 2. Comparisons between two data sets of mammal species occurrence. Species listed are on either Newmark's (1986) list of extant park mammals or on the park list but not both. Data are for Lassen Volcanic National Park and Lava Beds National Monument.

Lava Beds NM	**Lassen Volcanic NP**
Newmark's List	Newmark's List
Sorex vagrans	*Myotis volans*
	Spermophilus beecheyi
Park List	Park List
Myotis californicus	
Myotis yumanensis	*Lasionycteris noctivagans*
Myotis lucifugus	*Lasiurus cinerea*
Myotis volans	*Sylvilagus nuttallii*[b]
Myotis thysanodes	*Lepus californicus*[b]
Lasionycteris noctivagans	*Eutamias merriami*
Lasiurus cinerea	*Chaetodipus californicus*[a]
Eutamias minimus	*Castor canadensis*[b]
Ondatra zibethicus	*Microtus montanus*
Urocyon cinereoargenteus[b]	*Bassariscus astutus*[b]
Ursus americanus[b]	*Mustela erminea*[b]
Mustela vison[b]	*Mustela vison*[b]
Spilogale gracilis[b]	*Spilogale gracilis*

[a] Listed as *Perognathus* by the park.

[b] Species listed as extinct by Newmark.

may be too high for the black-tailed jackrabbit (sightings may be limited to dispersals from outside populations), the other species are known to be either common in the park, or there are records of roadkills and collections in the last few years (A. Denniston and R. C. Long, pers. comm.). The mink supposedly had disappeared before park establishment according to Newmark (1986), although the species is still on the park list.

Newmark (1986) also states that four species are extinct in Lava Beds NM (gray fox, spotted skunk, black bear, and mink). He reports no sighting records for gray fox and spotted skunk after 1973, nor for mink and black bear after park establishment in 1925. However, both spotted skunk and mink are known to occur in the park, and the black bear and gray fox are considered rare (B. Carter, in lit.).

As another example, Newmark (1987) cites a number of extinctions from Yosemite NP and Sequoia and Kings Canyon National Parks which cover 3,080 and 3,430 km^2, respectively. He apparently found no records for red fox in Sequoia and Kings Canyon following 1958 and none for striped skunk since 1973. Both species are reported to be present in the park and adjacent national forests (D. Graber and R.C. Long, pers. comm.). The most recent records found in Yosemite for the brush rabbit (*Sylvilagus bachmani*) and the red fox were 1957, and 1972 for the mink, while these species are considered extant in the park and are known from adjacent national forests (D. Graber and R. C. Long, pers comm.).

Alternate Species Lists

There are discrepancies among the alternate species lists that we compared for several parks, although all of these lists were supposedly drawn from park records. These discrepancies cannot be explained by differences in dates of assembly or revisions (compare dates of each study with dates of park lists in Table 1 or, where dates are unknown, the year we received the lists—1985). The species list from Lava Beds NM contains an additional 13 species not included on Newmark's (1986) list of extant mammals for this park (Table 2). Nine of the 13 species are those for which Newmark apparently found no records of occurrence past or present. Newmark's species list for Lassen Volcanic NP contains two species not included on the park list, and the park list contains six species which Newmark's does not. Some of the accounts reported by Van Gelder (1982) also disagree with park lists. According to Van Gelder, the bushy-tailed woodrat (*Neotoma cinerea*) and the golden-mantled ground squirrel (*Spermophilus lateralis*) have never been recorded in Redwood NP, though these species are on the park list. Redwood's list is derived from range inferences as well as empirical accounts, but the differences are not clearly distinguished. Van Gelder (1982) also states that the long-eared chipmunk (*Tamias quadrimaculatus*) is found in Yosemite at higher elevations, yet this species is not included in either Newmark's (1986) or the park's list.

Park Mammal Diversity and a Preliminary Wildlife Habitat Relations Analysis

Many of the park mammal lists are not complete (Table 1). A preliminary analysis of species distributions in conjunction with habitat preferences, using the wildlife habitat relations model of Bernard and Brown (1977) (based upon Kuchler's (1975) vegetation associations and physiographic region of occurrence), suggests some large errors in the species lists of several parks. The most obvious results are that two orders of mammals are not represented on several park lists. Four parks do not include any species of Insectivora on their lists, although all parks are predicted to contain species of this order. Chiroptera are predicted to occur in 15 of the 16 parks but only nine park lists include any bat species. This analysis also suggests that many parks may contain a larger number of these small mammal species, including many more rodent species than is presently known.

DISCUSSION

Comparability of Species Lists

The makeup of species lists is influenced by biological uncertainties concerning the status of species in the parks, as well as the purpose and anticipated uses of the lists. There are few published lists for mammals in California's parks, and some parks have multiple species lists, assembled at different times for different reasons. The rationale behind the different kinds of species lists and their anticipated use by the parks is understandable. However, these lists are generally not suitable for any sort of comparative species diversity analysis. Still, most of the lists contain valuable information on mammalian distributions in the parks. These lists could be very useful if the source and reliability of all list inclusions were identified and stored in a centralized location, and consistently updated with new information.

With regard to taxonomic standardization, changing taxonomies over time or disagreement among mammalogists can be a problem in resolving particular species identities when the taxonomic reference used is unknown. For example, following Jones et al. (1986), *Dipodomys pacificus* is a species that occurs over the same range as *Dipodomys heermani*, although *D. pacificus* is considered a subspecies of *D. heermani* by some taxonomists. The same is true for a number of other species. By maintaining species occurrence data in a centralized location, we could avoid the time and potential errors involved in standardizing species identity. All new reports could be easily compared with a computerized synonymy list which could also be useful for resolving possible misidentifications.

Confusion Over Occurrence

Many park lists do not reflect all the information concerning species occurrence and status that has been collected within the parks, both in historic and more recent times. Much of this information, in the form of documents, published papers, student theses, museum records and visitor card files, is often widely scattered in various locations. Therefore, when independent researchers attempt to assemble this information, the result can be numerous discrepancies between lists. For example, 23 and 25% of the species on Lassen and Lava Bed's park lists, respectively, were not included on Newmark's lists of species occurrence in these parks. Both park lists, however, are reportedly comprised of specimens and verified sightings.

That Newmark (1987) found "evidence" for so many post-establishment extinctions within the 14 parks he studied may be explained in part not only by a lack of thoroughness, but also by the state of many park records and methods of record-keeping. Newmark assumed extinction if he could find no evidence of a species occurring in a park within a 10-year period. However, we were able to disavow these apparent extinctions by contacting certain knowledgeable individuals. This example demonstrates that records of species occurrence are too often not kept in a central, easily accessible location for researchers. Furthermore, records of many commonly observed species, such as rabbits and skunks, are not consistently kept. One contributing factor to this problem is that the card system, formerly used by park rangers for recording species encountered in the parks, has for the most part fallen out of favor in recent years.

On the other hand, because some park lists may not be periodically updated, they may contain entries of species that are truly extinct. Examples are the mink at Lassen and the black bear at Lava Beds, which went extinct prior to their establishment (Newmark, 1987).

Although most of the currently available species lists to not represent true inventories, they may be useful to some degree if more information could be assembled regarding the criteria used to include and exclude species for listing. These criteria include:

1. How well do the lists represent the current knowledge of mammal occurrence in the parks?
2. How reliable are the inclusions (e.g., are there verified sightings or specimens)?
3. The dates of the records;
4. The taxonomic classifications used in the species identifications;

5. The number of sightings of each species recorded; and
6. Any empirical or anecdotal information concerning species status.

Additional Information to be Gathered and the Implications of a Wildlife Habitat Relations Analysis

Few parks presently have the monetary or staffing resources needed to gather adequate mammal inventory information. However, a significant amount of information concerning species occurrences, not known to the parks, is available in the literature. Although our study was far from extensive, we found reports that were not registered by Newmark (1986) or Van Gelder (1982). For example, we found a "marginal" record for Merriam's shrew at Lava Beds NM in Hall (1981). Miller and Stebbins (1964) describe taking specimens of *Reithrodontomys megalotis* and *Microtus californicus* and finding raccoon tracks at the western border of Joshua Tree NM. Much knowledge may also be held by experts that has not been published. During the course of our study, we were informed that *Myotis evotis* is present at Death Valley NM (G. Fellers, in lit.), and that gray fox is present at Lassen (A. Denniston, pers. comm.). These are all additions to the park lists. Gathering this information together could contribute greatly to our knowledge of mammalian diversity in the national parks of California.

Finally, a preliminary analysis of mammalian species occurrence using a wildlife habitat relations (WHR) model suggests that mammalian species richness may be greater within California's national parks than previously reported (California Department of Fish and Game, 1988). Examination of the reported diversity within parks, with regard to the predictions of occurrence based upon range and habitat preference, shows that four parks list no species of insectivores, four list no species of bats, two no murid rodents, and one no heteromyid rodents. Many parks potentially harbor a larger number of species than are represented by their extant mammal lists.

Thirty-two of California's native terrestrial mammal species are not on any of the 16 park lists. Several of these species are listed as rare, threatened or in danger of extinction by the State of California. Several are California endemics, or have endemic subspecies. WHR models may be used to predict the occurrences of species previously not known to exist in a park. Given the traditional scarcity of funds for conducting thorough inventories of park biota, these models may be useful as a first step towards completing the faunal inventories of many parks. In addition, comparable knowledge of the distribution and occurrence of particularly obscure species, such as many insectivores, bats, and rodents, may be our only means for assessing the conservation effectiveness of reserves.

CONCLUSIONS

Regained enthusiasm for research in the national parks (Sumner, 1983), recent calls for a national biological survey (Wilson, 1985) and concern for the rapid depletion of habitat and extinction of species indicate that the time is right for a thorough assessment of biological inventory data for the national parks. A central, standardized system for recording and maintaining faunal diversity data for the national parks should be established as soon as possible. Specific recommendations for this system, with regard to mammals, are as follows:

1. Taxonomic identifications of all currently listed entries should be verified to the extent possible and standardized to Jones et al. (1986). This classification system should be updated periodically for wide use by mammalogists. It is also used by the California Wildlife Habitat System (California Department of Fish and Game, 1988).
2. The NPS mammal database for California parks should be compatible with modern natural resource information networks on regional (such as the GIS in development by the State of California) and national levels.
3. The quality of all currently available data should be assessed. In particular, where possible, information should be gathered for all park species lists with regard to: the criteria used to include and exclude species from park lists; how well the lists represent the current knowledge of mammal occurrence in the parks; the reliability of inclusions; dates of the records; taxonomic classifications used to identify species; subspecific information; number of sightings and the dates of these; and any empirical or anecdotal information concerning species status. This information should eventually be incorporated into the database.
4. All reference materials that may contain information on species occurrence in the parks, both current and historical, should be compiled. National and regional museums should be surveyed for park specimens and records, and experts should be consulted for their knowledge of any species occurrences. This information should be added to the database.
5. All future reporting of species sightings should include standard information compatible with a GIS (see introductory chapter), including verification of the sighting, location, habitat type and condition, slope, aspect, and weather conditions.

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APPENDIX 1.* Park by park accounting of mammal species occurrence data contained in the UC/CPSU database. Data are species lists provided by the parks with recent updates and verifications. For source of data and comments, see Table 1.

Species lists for 16 National Park Service units in California. Parks are listed in order of increasing size. Species present on park lists are indicated by "1", not on lists by "0". All data were provided by the parks (Table 1).

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Didelphis virginiana Opposum	0	0	0	1	0	1	0	1	0	1	0	1	0	0	1	0
Sorex lyelli Mount Lyell Shrew	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sorex vagrans Vagrant Shrew	0	0	0	0	0	1	0	0	0	1	0	0	0	0	1	0
Sorex monticolus Dusky (Montane) Shrew	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sorex pacificus Pacific Shrew	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Sorex ornatus Ornate Shrew	0	0	0	1	0	1	0	0	0	0	0	1	0	0	1	0
Sorex tenellus Inyo Shrew	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sorex palustris Water Shrew	0	0	1	0	0	0	0	0	1	1	0	0	0	1	1	0
Sorex bendirii Pacific Water Shrew	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sorex trowbridgii Trowbridge's Shrew	0	0	0	0	0	1	1	1	1	1	0	0	0	0	1	0
Sorex merriami Merriam's Shrew	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI^c	DEVA
Notiosorex crawfordi Desert Shrew	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Neurotrichus gibbsii Shrew-mole	0	0	0	0	0	1	0	1	1	1	0	0	0	0	0	0
Scapanus townsendii Townsend's Mole	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Scapanus orarius Coast Mole	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Scapanus latimanus Broad-footed Mole	0	0	0	1	0	1	1	1	0	1	0	1	0	0	1	0
Macrotus californicus California Leaf-nosed Bat	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Choeronycteris mexicana Mexican Long-tongued Bat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Myotis lucifugus Little Brown Myotis	0	0	0	0	0	1	1	1	1	1	0	0	0	0	1	0
Myotis yumanensis Yuma Myotis	0	0	0	0	0	1	1	1	1	0	1	0	1	0	1	0
Myotis velifer Cave Myotis	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Myotis evotis Long-eared Myotis	0	0	0	1	0	1	1	1	1	1	1	0	1	0	1	1
Myotis thysanodes Fringed Myotis	0	0	0	1	0	1	1	1	1	0	0	0	1	0	0	1
Myotis volans Long-legged Myotis	0	0	0	0	0	1	1	1	1	0	1	0	1	0	1	0
Myotis californicus California Myotis	0	0	0	1	0	1	1	1	1	1	1	0	1	0	1	1

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Myotis ciliolabrum Western Small-footed Myotis	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	1
Lasionycteris noctivagans Silver-haired Bat	0	0	0	0	0	1	1	1	1	1	0	0	0	0	1	1
Pipistrellus hesperus Western Pipistrelle	0	0	0	1	0	1	0	1	1	0	0	0	1	0	1	1
Eptesicus fuscus Big Brown Bat	0	0	0	1	0	1	1	1	1	1	1	0	1	0	1	0
Lasiurus borealis Red Bat	0	0	0	1	0	1	0	1	1	0	0	0	1	0	1	0
Lasiurus cinerea Hoary Bat	0	0	0	0	0	1	1	1	1	1	0	0	1	0	1	1
Lasiurus ega Southern Yellow Bat	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Euderma maculatum Spotted Bat	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0
Plecotus townsendii Townsend's Big-eared Bat	0	0	0	1	0	1	1	1	1	0	1	0	1	0	1	1
Antrozous pallidus Pallid Bat	0	0	0	0	0	1	0	1	0	0	1	0	1	0	1	1
Tadarida brasiliensis Brazilian Free-tailed Bat	0	0	0	0	0	1	1	1	1	0	1	0	1	0	1	1
Tadarida femorosacca Pocketed Free-tailed Bat	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Tadarida macrotis Big Free-tailed Bat	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Eumops perotis Western Mastiff Bat	0	0	0	1	0	1	0	0	0	0	0	0	1	0	0	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Ochotona princeps Pika	0	0	0	0	0	0	1	0	0	1	0	0	0	1	1	0
Brachylagus idahoensis Pygmy Rabbit	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sylvilagus bachmani Brush Rabbit	1	1	0	1	0	1	0	1	1	0	0	1	0	1	1	0
Sylvilagus nuttalli Nuttall's (Mountain) Cottontail	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	1
Sylvilagus audubonii Audubon's (Desert) Cottontail	0	0	0	1	0	1	0	0	0	0	0	1	1	0	1	1
Oryctolagus cuniculus[b] Europena Rabbit	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Lepus americanus Snowshoe Hare	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Lepus townsendii White-tailed (Hare) Jackrabbit	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0
Lepus californicus Black-tailed (Hare) Jackrabbit	0	0	0	1	0	1	1	1	0	1	0	0	1	1	1	1
Aplodontia rufa Mountain Beaver	0	0	0	0	0	1	0	1	1	1	0	0	0	0	1	0
Tamias alpinus Alpine Chipmunk	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0
Tamias minimus Least Chipmunk	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0
Tamias amoenus Yellow-pine Chipmunk	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0
Tamias townsendii Townsend's Chipmunk	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Tamias sonomae Sonoma Chipmunk	0	1	0	0	0	1	0	1	0	0	0	0	0	0	0	0
Tamias merriami Merriam's Chipmunk	0	0	0	1	0	1	1	0	0	1	0	0	1	0	1	0
Tamias obscurus California Chipmunk	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tamias quadrimaculatus Long-eared Chipmunk	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tamias speciosus Lodgepole Chipmunk	0	0	1	0	0	0	0	0	0	1	0	0	0	1	1	0
Tamias panamintinus Panamint Chipmunk	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Tamias umbrinus Uinta Chipmunk	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Marmota flaviventris Yellow-bellied Marmot	0	0	1	0	0	0	1	0	0	1	0	0	0	1	1	0
Ammospermophilus leucurus White-tailed Antelope Squirrel	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Ammospermophilus nelsoni San Joaquin (Nelson's) Antelope Squirrel	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Spermophilus townsendii Townsend's Ground Squirrel	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Spermophilus beldingi Belding's Ground Squirrel	0	0	1	0	0	0	1	0	0	1	0	0	0	1	1	0
Spermophilus variegatus Rock Squirrel	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Spermophilus beecheyi California Ground Squirrel	1	0	0	1	0	1	1	0	1	0	0	1	1	1	1	1
Spermophilus mohavensis Mohave Ground Squirrel	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Spermophilus tereticaudus Round-tailed Ground Squirrel	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Spermophilus lateralis Golden-mantled Ground Squirrel	0	0	1	0	0	0	1	0	1	1	0	0	0	1	1	0
Sciurus carolinensis[b] Gray Squirrel	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sciurus griseus Western Gray Squirrel	0	1	0	0	1	1	0	1	1	1	0	1	0	1	1	0
Sciurus niger[b] Fox Squirrel	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Tamiasciurus douglasii Douglas' Squirrel	0	0	1	0	0	1	1	0	1	1	0	0	0	1	1	0
Glaucomys sabrinus Northern Flying Squirrel	0	0	0	0	0	0	0	0	1	1	0	0	0	1	1	0
Thomomys bottae Botta's Pocket Gopher	0	1	0	1	0	1	0	1	1	0	0	1	1	1	1	1
Thomomys townsendii Townsend's Pocket Gopher	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Thomomys talpoides Northern Pocket Gopher	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Thomomys mazama Western Pocket Gopher	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Thomomys monticola Mountain Pocket Gopher	0	0	1	0	0	0	0	0	0	1	0	0	0	1	1	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Perognathus longimembris Little Pocket Mouse	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Perognathus inornatus San Joaquin Pocket Mouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Perognathus parvus Great Basin Pocket Mouse	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1
Perognathus alticola White-eared Pocket Mouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Perognathus xanthonotus Yellow-eared Pocket Mouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chaetodipus formosus Long-tailed Pocket Mouse	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Chaetodipus baileyi Bailey's Pocket Mouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chaetodipus penicillatus Desert Pocket Mouse	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Chaetodipus fallax San Diego Pocket Mouse	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Chaetodipus californicus California Pocket Mouse	0	0	0	1	0	1	0	0	0	1	0	0	0	0	1	0
Chaetodipus spinatus Spiny Pocket Mouse	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Microdipodops megacephalus Dark Kangaroo Rat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Microdipodops pallidus Pale Kangaroo Rat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Dipodomys ordii Ord's Kangaroo Rat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dipodomys microps Chisel-toothed Kangaroo Rat	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Dipodomys elephantinus Big-eared Kangaroo Rat	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Dipodomys venustus Narrow-faced Kangaroo Rat	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Dipodomys agilis Pacific (Agile) Kangaroo Rat	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Dipodomys heermanni Heermann's Kangaroo Rat	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Dipodomys californicus California Kangaroo Rat	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0
Dipodomys ingens Giant Kangaroo Rat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dipodomys panamintinus Panamint Kangaroo Rat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Dipodomys stephensi Stephens' Kangaroo Rat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dipodomys deserti Desert Kangaroo Rat	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Dipodomys merriami Merriam's Kangaroo Rat	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Dipodomys nitratoides San Joaquin (Fresno) Kangaroo Rat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Castor canadensis Beaver	0	0	0	0	0	0	0	0	1	1	0	0	0	0	1	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Reithrodontomys megalotis Western Harvest Mouse	0	0	0	1	0	1	1	0	1	0	1	0	1	1	1	1
Reithrodontomys raviventris Salt-marsh Harvest Mouse	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Peromyscus eremicus Cactus Mouse	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Peromyscus californicus California Mouse	0	0	0	1	0	1	0	0	0	0	0	1	0	0	1	0
Peromyscus maniculatus Deer Mouse	0	1	1	1	0	1	1	1	1	1	1	1	1	1	1	1
Peromyscus crinitus Canyon Mouse	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	1
Peromyscus boylii Brush Mouse	0	0	0	1	0	1	0	0	0	0	0	1	1	0	1	1
Peromyscus truei Pinon (Pinyon) Mouse	0	0	0	1	0	1	1	0	1	0	0	1	1	0	1	1
Onychomys leucogastor Northern Grasshopper Mouse	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Onychomys torridus Southern Grasshopper Mouse	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Sigmodon hispidus Hispid Cotton Rat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sigmodon arizonae Arizona Cotton Rat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Neotoma albigula White-throated Woodrat	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Neotoma lepida White-throated (Desert) Woodrat	0	0	0	1	0	0	0	0	0	0	0	1	1	0	0	1
Neotoma fuscipes Dusky-footed Woodrat	0	0	0	1	0	1	1	1	1	0	0	0	1	1	1	0
Neotoma cinerea Bushy-tailed Woodrat	0	0	0	0	0	0	1	0	1	1	0	0	0	1	1	1
Clethrionomys californicus Western Red-backed Mouse	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0
Phenacomys intermedius Heather Vole	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Phenacomys albipes White-footed Vole	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Phenacomys longicaudus Reddish Tree Mouse	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0
Microtus montanus Montane Vole	0	0	0	0	0	0	1	0	0	1	0	0	0	0	1	0
Microtus californicus California Vole	0	0	0	1	0	1	0	1	1	0	0	1	1	0	1	0
Microtus townsendii Townsend's Vole	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Microtus longicaudus Long-tailed Vole	0	0	0	0	0	0	0	0	1	1	0	0	0	0	1	0
Microtus oregoni Creeping Vole	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Lemmiscus curtatus Sagebrush Vole	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ondatra zibethicus Muskrat	0	0	0	0	0	1	1	1	1	1	0	0	0	0	0	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Rattus rattus[b] Black Rat	0	0	0	0	0	1	0	0	1	0	1	0	0	0	0	0
Rattus norvegicus[b] Norway Rat	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	0
Mus musculus[b] House Mouse	0	0	0	0	0	1	0	1	1	0	0	0	1	0	0	1
Zapus princeps Western Jumping Mouse	0	0	0	0	0	0	0	0	0	1	0	0	0	0	1	0
Zapus trinotatus Pacific Jumping Mouse	0	0	0	0	0	1	0	1	1	0	0	0	0	0	0	0
Erethizon dorsatum Porcupine	0	0	1	0	0	0	1	1	1	1	0	0	0	1	1	1
Canis latrans Coyote	0	0	1	1	1	1	1	1	1	1	0	1	1	1	1	1
Canis familiaris[a] [b] Dog	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0
Vulpes vulpes Red Fox	0	0	0	0	0	1	1	1	0	1	0	1	0	0	1	0
Vulpes macrotus Kit Fox	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Urocyon cinereoargenteus Gray Fox	1	0	0	1	1	1	1	1	1	1	0	1	1	1	1	1
Urocyon littoralis Island Fox	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Ursus americanus Black Bear	0	0	1	0	1	0	1	0	1	1	0	0	0	1	1	0
Callorhinus ursinus Northern Fur Seal	0	0	0	0	0	1	0	1	1	0	1	0	0	0	0	0

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APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Arctocephalus townsendi Guadelupe Fur Seal	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Eumetopias jubatus Northern Sea Lion	0	0	0	0	0	1	0	1	1	0	1	0	0	0	0	0
Zalophus californianus California Sea Lion	0	0	0	0	0	1	0	1	1	0	1	0	0	0	0	0
Bassariscus astutus Ringtail	0	0	0	1	1	1	0	1	1	1	0	1	1	1	1	1
Procyon lotor Raccoon	0	1	1	1	1	1	1	1	1	1	0	1	1	1	1	0
Martes americana Marten	0	0	1	0	0	0	0	0	1	1	0	0	0	1	1	0
Martes pennanti Fisher	0	0	0	0	0	0	0	0	1	0	0	0	0	0	1	0
Mustela erminea Ermine	0	0	0	0	0	1	0	1	1	1	0	0	0	0	1	0
Mustela frenata Long-tailed Weasel	0	0	1	1	0	1	1	1	1	1	0	1	1	1	1	0
Mustela vison Mink	0	0	0	0	0	1	1	0	1	1	0	0	0	1	1	0
Gulo gulo Wolverine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Taxidea taxus Badger	0	0	0	1	0	1	1	1	0	1	0	1	1	1	1	1
Spilogale gracilis Western Spotted Skunk	0	0	0	0	0	1	1	1	1	1	1	0	1	1	1	1
Mephitis mephitis Striped Skunk	0	0	0	1	0	1	1	1	1	1	0	0	1	1	1	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Lutra canadensis River Otter	0	0	0	0	0	1	0	0	1	0	0	0	0	1	0	0
Enhydra lutris Sea Otter	0	0	0	0	0	1	0	1	1	0	1	0	0	0	0	0
Phoca vitulina Harbor Seal	0	0	0	0	0	1	0	1	1	0	1	1	0	0	0	0
Histriophora fasciata Ribbon Seal	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mirounga angustirostris Northern Elephant Seal	0	0	0	0	0	1	0	1	1	0	1	0	0	0	0	0
Felis concolor Mountain Lion	0	0	0	1	1	1	1	1	1	1	0	1	1	1	1	1
Felis domesticus[a, b] Domestic Cat	0	0	0	0	0	1	0	1	0	0	1	1	0	0	1	0
Felis rufus Bobcat	0	0	0	1	1	1	1	1	1	1	0	1	1	1	1	1
Eschrichtius robustus Gray Whale	0	0	0	0	0	1	0	1	1	0	1	0	0	0	0	0
Balaenoptera acutorostrata Minke Whale	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Balaenoptera borealis Sei Whale	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Balaenoptera musculus Blue Whale	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Balaenoptera physalus Fin-backed Whale	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Megaptera novaeangliae Hump-backed Whale	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Balaena glacialis Black Right Whale	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0
Steno bredanensis Rough-toothed Dolphin	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Tursiops truncatus Bottle-nosed Dolphin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Stenella attenuata Pantropical Spotted Dolphin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Stenella coeruleoalba Striped Dolphin	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Delphinus delphis Common (Saddle-backed) Dolphin	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Lagenorhynchus obliquidens Pacific White-sided Dolphin	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Grampus griseus Grampus (Risso's Dolphin)	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Pseudorca crassidens False Killer Whale	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Globicephala macrorhynchus Short-finned Pilot Whale	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Orcinus orca Killer Whale	0	0	0	0	0	1	0	1	1	0	1	0	0	0	0	0
Lissodelphis borealis Northern Right-whale Dolphin	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Phocoena phocoena Harbor Porpoise	0	0	0	0	0	1	0	1	1	0	1	0	0	0	0	0
Phocoenoides dalli Dall's Porpoise	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Berardius bairdii North Pacific Bottle-nosed Whale	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Ziphius cavirostris Goose-beaked Whale	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Mesoplodon stejnegeri North Pacific (Bering Sea) Beaked Whale	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0
Mesoplodon ginkgodens Ginkgo-toothed Beaked Whale	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Mesoplodon carlhubbsi Moore's (Arch-beaked) Beaked Whale	0	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0
Mesoplodon densirostris Dense-beaked Whale	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mesoplodon hectori Hector's Beaked Whale	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Kogia breviceps Pygmy Sperm Whale	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Kogia simus Dwarf Sperm Whale	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Physeter macrocephalus Sperm Whale	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Equus caballus[a b] Feral Horse	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	1
Equus asinus[b] Feral Burro	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Sus scrofa[b] Feral Pig	0	0	0	0	1	1	0	0	1	0	1	0	0	0	1	0
Cervis elaphus Wapiti or Elk	0	0	0	0	0	0	0	1	1	0	1	0	0	0	0	0
Cervus dama[b] Fallow Deer	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0
Cervus unicolor[b] Sambar	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cervus axis[b] Axis Deer	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0
Capreolus pyargus[a b] Snow Deer	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Odocoileus hemionus Black-tailed Deer	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Antilocapra americana Pronghorn	0	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0
Bison bison Bison	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bos taurus[a b] Cattle	0	0	0	0	0	1	0	0	0	0	1	0	0	0	1	0
Ovis canadensis Mountain Sheep	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Ovis aries[a b] Sheep	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Ammotragus lervia[b] Barbary Sheep	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hemitragus jemlahicus[b] Himalayan Tahr	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

APPENDIX 1. (continued)

	CABR	MUWO	DEPO	PINN	WHIS	GOGA	LABE	PORE	REDW	LAVO	CHIS	SAMO	JOTR	YOSE	SEKI[c]	DEVA
Capra hircus[b] Feral Goat	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

* Data for this appendix were compiled from existing lists provided by the parks and the following published material: Fellers and Arnold (1987); NPS, 1975; NPS, 1985.

[a] Not listed by Jones et al. (1986).

[b] Exotic or domestic species.

[c] Graber, in prep.

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Monitoring the Population Dynamics of a Reintroduced Mountain Sheep Herd in the Sierra Nevada, California

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Abstract. We monitored a reintroduced herd of mountain sheep (*Ovis canadensis californiana*) from March 1986 to March 1988. Study objectives included documenting reproduction, survival, and sources of mortality. Lamb:ewe ratios in 1986 and 1987 were 75:100 and 83:100 and lamb survival to one year of age was 67% and 60%, respectively. Confirmed mortality for the first year was 33%, but actual mortality was probably higher (44%). During the second year, confirmed mortality declined to 10% while actual mortality was postulated at 20%. Primary sources of confirmed mortality were puma (*Felis concolor*) predation (43%) and exposure to severe post-release weather (29%).

INTRODUCTION

Mountain sheep originally occupied the Sierra Nevada wherever suitable rocky terrain with access to wintering areas existed (Jones, 1950; Wehausen, 1980). Population numbers rapidly declined in the late 1800s, apparently from overhunting, competition for forage with domestic sheep, and diseases contracted from domestic stock (Buechner, 1960; Wehausen, 1980). Distribution in the Sierra is currently limited to six populations. Native herds of this subspecies have survived at Mt. Baxter, Sawmill Creek and Mt. Williamson (Wehausen, 1980; C. D. Hargis, unpubl. memo to the Sierra Nevada Bighorn Sheep Interagency Advisory Group, Inyo National Forest; Lee Vining, 1986). Reintroductions of native mountain sheep have been made at Wheeler Crest (Andaloro and Ramey, 1981), Mt. Langley, and Lee Vining Canyon (Keay et al., 1987).

On 5 and 6 March 1986, 27 California mountain sheep were captured on the eastern slopes of Mt. Baxter, Inyo County, California and transplanted 120 km north to Lee Vining Canyon, Mono County (Keay et al., 1987). The Lee Vining reintroduction was the third such reintroduction in a long-term program to reestablish geographically disjunct populations of mountain sheep on historic ranges in the Sierra Nevada (Sierra Nevada Bighorn Sheep Interagency Advisory Group, 1984).

To evaluate the Lee Vining reintroduction's success or failure, the National Park Service initiated a three-year research study at the time of release. Primary objectives of the study were to: (1) document natality and mortality within the new herd; (2) determine seasonal patterns of distribution and range use; and (3) investigate the effects of habitat on survival and reproduction. Here, we report herd natality and mortality in the first two years following reintroduction.

METHODS

Study Area

The 46 km^2 study area is in the central Sierra Nevada of California, 5 km east of Yosemite National Park. It is bounded by Lundy Canyon on the north and Lee Vining Canyon on the south, and lies within Inyo National Forest. Elevation ranges from 2188 m at the eastern end of Lee Vining Canyon to 3758 m atop the summit of Mt. Warren.

Weather on the study area is characterized by cold, wet winters and warm, dry summers. Annual precipitation averages 69 cm per year and falls primarily as snow deposited between November and April.

Lower elevations used by mountain sheep as winter ranges (2180-2590 m) are characterized by sagebrush-steppe vegetation (Major, 1977). Sheep summer ranges include two distinct plant communities: subalpine forest and alpine fell-field (Munz and Keck, 1959).

Radio transmitter collars (Telonics, Mesa, AZ) equipped with mortality sensors were installed on 25 of the 27 transplanted sheep at the time of their release. Each sheep's survival was subsequently monitored five to seven times weekly using radio telemetry. We conducted censuses of all collared sheep approximately every 10 days by hiking to areas of use and viewing the sheep with 10 × 25 binoculars and a 15-45× spotting scope. Because lambs and yearlings remain with ewe groups through their second year (Geist, 1971), we were able to document lamb recruitment and the survival of uncollared sheep.

Mortality sensors are activated after a radio collar has remained stationary for six hours. When a mortality signal was received, we attempted to locate the collar and its associated animal, usually within 24 hours. If a dead sheep was found, we performed a necropsy to determine the cause of death. We assumed that an uncollared sheep had died if, after three months, we repeatedly failed to see it while censusing the collared animals. In such cases, cause of death could not be determined.

RESULTS

Survival and Reproduction

The composition of the original herd transplanted to Lee Vining Canyon was 13 adult ewes, seven adult rams, and seven lambs (three females, four males). The sheep were released in two groups on successive days. After one year, 14 of 27 sheep from the initial transplant were still alive. This number declined to 12 by the end of the second year. Three of the surviving sheep were ewes that, in September 1986, migrated and established residence in Bloody Canyon, 13 km south of Lee Vining. Of the nine transplanted sheep remaining in Lee Vining Canyon after two years, four were adult rams and five were adult ewes.

Between 20 April and 20 May 1986, nine of 12 adult ewes transplanted to Lee Vining bore lambs, yielding a spring lamb:ewe ratio of 75:100. In 1987, five of six adult ewes remaining in Lee Vining Canyon bore lambs. Because no adult rams visited Bloody Canyon during the breeding season, none of the ewes there produced a lamb in 1987. Thus, the spring 1987 lamb:ewe ratio was 83:100 for Lee Vining Canyon and 56:100 overall.

Six of the nine lambs born in 1986 survived to two years of age. Of these, four have remained in Lee Vining Canyon. Three are males and one is a female. The other two surviving 1986 lambs, a male and a female, migrated with their ewes to Bloody Canyon in the autumn of 1986. Three of the five lambs born in 1987 have survived to the present, and are females. Lamb survival to one year of age in 1987 and 1988 was 67% and 60%, respectively.

The Lee Vining herd contained 21 sheep two years after release (Table 1). Twelve were from the original reintroduction and nine were born there. Sixteen sheep (five adult ewes, four adult rams, three male and one female two-year-olds, and three female yearlings) established residence in Lee Vining Canyon. Five sheep—three adult ewes and a two-year-old of each sex—dispersed to Bloody Canyon.

TABLE 1. Location, age class, sex, and location of surviving mountain sheep, two years after reintroduction to Lee Vining Canyon, California as of 31 March 1988.

Location	Lambs		Yearlings		Adults (>2 years old)	
	M	F	M	F	M	F
Lee Vining Canyon	0	3	3	1	4	5
Bloody Canyon	0	0	1	1	0	3

Mortality

Twenty sheep died in the two years following the reintroduction (Table 2). A third (n = 7) of the mortality occurred within three weeks of the release and primarily involved sheep that moved to higher elevations (> 3200 m) after their release on the second day. Among these were two adult rams and an adult ewe. Four of the five radio-collared lambs died while the two uncollared lambs released with the initial transplant have not been seen and are presumed dead. We recovered the carcasses of all seven radio-collared sheep.

It snowed daily for two weeks following the original transplant and average daytime temperatures were –8°C. Two sheep died when they fell into snow-covered crevices and became trapped. Necropsies of the five remaining carcasses found evidence of nutritional stress. Fat reserves were severely depleted in four cases and weight loss was as high as 28% in lambs. Rumen contents included undigested twigs and pine needles. Examinations of muscle tissue yielded no evidence of capture myopathy for most of the dead sheep. These findings suggest that the majority of post-release mortality (n = 4) resulted from malnutrition and exposure (L. Chow, unpub. data).

TABLE 2. Sources of mortality in the Lee Vining herd between 6 March 1986 and 31 March 1988.

Cohort	No.	Sources of Mortality				
		Exposure	Accident	Predation	Unknown	Total
Transplant	27	4	2	5	4	15
1986 lambs	9	0	0	0	3	3
1987 lambs	5	0	0	1	1	2
Total	41	4	2	6	8	20

There were no mortalities during the spring and summer. Between 1 September 1986 and 31 March 1987, three ewes and one ram were lost to puma (*Felis concolor*) predation. Three 1986 lambs died during the winter. One of them was found dead at the base of a cliff and two others disappeared. The causes of these deaths could not be determined.

Confirmed mortality for the first year was 33% (n = 12), although actual mortality was probably 44% (n = 16). Of the known causes of mortality, exposure accounted for 40% (n = 4) while predation and accidents accounted for 40% (n = 4) and 20% (n = 2), respectively.

Four of the 20 sheep present in Lee Vining Canyon on 1 May 1987 had died by 31 March 1988 while none died in Bloody Canyon. Two of the mortalities, an adult ewe and a female lamb, were due to puma predation. A lamb and a two-year-old ram were last seen on 2 November 1987 and 30 January 1988, respectively, and are presumed dead. Thus, confirmed mortality in 1987 was 10% (n = 2) although actual mortality was more likely 20% (n = 4).

DISCUSSION

Intensive monitoring of the Lee Vining herd after reintroduction has allowed us to determine natality and mortality rates within the new herd. Monitoring has also enabled us to document causes of mortality so that steps can be taken to reduce future losses.

With the exception of those sheep that died immediately after the transplant, puma predation accounted for six of the seven deaths in which we have been able to examine the carcass. The high percentage of puma predation among dead radio-collared sheep suggests the likelihood that predation may also be responsible for the disappearance of some of the uncollared sheep.

Eighty-three percent (n = 5) of the predation has been confined to the winter months. At that time, snow at higher elevations forces the herd onto its winter range in lower Lee Vining Canyon (Chow et al., 1988). Sierran mountain sheep usually summer in alpine areas (Wehausen, 1980) where visibility and the ability to detect predators is unimpaired by dense vegetation. The brushy and often broken habitat found on the winter range may impair the sheeps' detection of predators, thereby rendering the sheep more vulnerable to predation.

Lamb:ewe ratios for the first two years following the reintroduction were comparable to those of rapidly growing mountain sheep populations (Buechner, 1960) and were twice the figure given by Wehausen (1983) as necessary for maintenance of the Mt. Baxter herd. Despite a high birth rate in the Lee Vining herd, population growth has been limited by emigration and mortality. These factors were considered to be of negligible importance in the populations examined by Buechner (1960).

Holling (1959) found that predators can play an important role in population regulation when prey numbers are low but become less important when prey numbers are high. These findings might apply to the present situation in Lee Vining Canyon, where continued predation on the small sheep population may render the herd inviable and lead to local extinction. Despite high lamb:ewe ratios, the herd's rate of population growth will be severely limited by the low number of reproductive ewes remaining in the population. Further losses of reproductive age ewes is likely to seriously compromise the success of the reintroduction.

CONCLUSIONS

Management implications of these preliminary data include: (1) excluding lambs from future transplants because of their greater susceptibility to mortality in an unfamiliar environment; (2) predator control in the early years may be necessary to protect the breeding potential of the population; and (3) supplementation of the Lee Vining Herd with additional reproductive age ewes will offset predation losses with a higher rate of population growth. Any additional ewes released in Lee Vining Canyon should originate from the Mt. Baxter herd so that the genetic integrity of Sierran sheep is maintained until taxonomic issues have been resolved.

ACKNOWLEDGEMENTS

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Food Habits of Fishes in the Redwood Creek Estuary

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Abstract. The food habits of 11 fish species in the Redwood Creek estuary and an adjacent stream and sloughs were studied during 1980 to identify organisms consumed, seasonal and spatial dietary patterns, and levels of dietary overlap. Fish examined included anadromous species: chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*), and steelhead trout (*O. mykiss*); marine species: starry flounder (*Platichthys stellatus*), surf smelt (*Hypomesus pretiosus*), and shiner surfperch (*Cymatogaster aggregata*); freshwater species: prickly scuplin (*Cottus asper*), Sacramento sucker (*Catostomus occidentalis*), and threespine stickleback (*Gasterosteus aculeatus*); and euryhaline species: staghorn scuplin (*Leptocottus armatus*) and tidewater goby (*Eucyclogobius newberryi*). Immature aquatic insects, especially dipterans, were most important in the diets of most fish species. Aquatic crustaceans, particularly two species of the benthic amphipod, *Corophium*, were very important in the summer diet of juvenile steelhead and prickly sculpin in the estuary. Because the diets of the different fish species tended to emphasize different species of prey, dietary overlap was biologically insignificant. Predation of fish on other fish was found to be generally of little importance.

INTRODUCTION

Estuaries of the northwestern United States have been identified as important feeding areas for a variety of fishes, particularly juvenile salmonids (Snyder, 1931; Needham, 1940; Reimers, 1973; Neilson et al., 1985). These estuaries may support, in addition to anadromous species, marine, freshwater, and euryhaline species. Developments within, or modifications to, an estuary may affect its productivity. In 1980 we studied the food habits of fish found in the Redwood Creek estuary, Humboldt

County, California, to better understand the importance of this estuary to the production of fish (for full details see Salamunovich, 1987). We wished to determine the species of fish utilizing the estuary, the seasonal or spatial patterns in their diets, and to compare their diets.

STUDY AREA

Redwood Creek flows through the Redwood National Park and enters the Pacific Ocean approximately 80 kilometers south of the California-Oregon border (Fig. 1). The stream drains approximately 73,000 hectares (Iwatsubo et al., 1976) and has summer flows averaging 1.2 m^3/s (U.S. Geological Survey, 1980, 1981). The stream has historically supported substantial runs of chinook salmon, coho salmon, and steelhead (Ricks and Feranna, 1981).

Because of flood damages, particularly to the community of Orick located about four kilometers upstream from the mouth of Redwood Creek, levees were constructed in 1968 along the lower five kilometers of the stream by the U.S. Army Corps of Engineers (Fig. 2). These levees substantially altered the estuarine habitat by removing riparian vegetation and a major meander of the stream. These alterations to the estuarine habitat, along with logging, road building and grazing, may have contributed to declines in local runs of anadromous salmonids (Winzler and Kelley, 1970; U.S. Fish and Wildlife Service, 1975; Larson et al., 1983).

The character of the Redwood Creek estuary varies seasonally (Gregory, 1982). Salt water intrusion into the estuary is limited in the winter and spring by high stream flows, while low summer and fall flows permit the formation of a sand berm across the mouth of the creek that frequently blocks the interchange of fresh and salt water. During this study, the berm was breached in early July, discharging fish from the embayment into the ocean and disturbing the subtrates and benthic fauna in the embayment (Larson et al., 1982, 1983; Larson, 1987).

METHODS

We studied the food habits of fish in lower Redwood Creek throughout 1980 in four separate habitats: (1) the estuary; (2) the stream below the U.S. Highway 101 bridge and immediately above the estuary; (3) the South Slough, which is the remnant of a former stream meander; and (4) the North Slough, which is a former tributary channel with periodic access to the estuary (Fig. 2).

Most fish were collected by seining, with a few caught in a minnow trap. Juvenile stickleback were also collected in dredge and plankton samples. Because the size range of the fish was relatively small, we do not feel that there was any particular selectivity of the fish caught. No effort was made to collect adult anadromous salmonids.

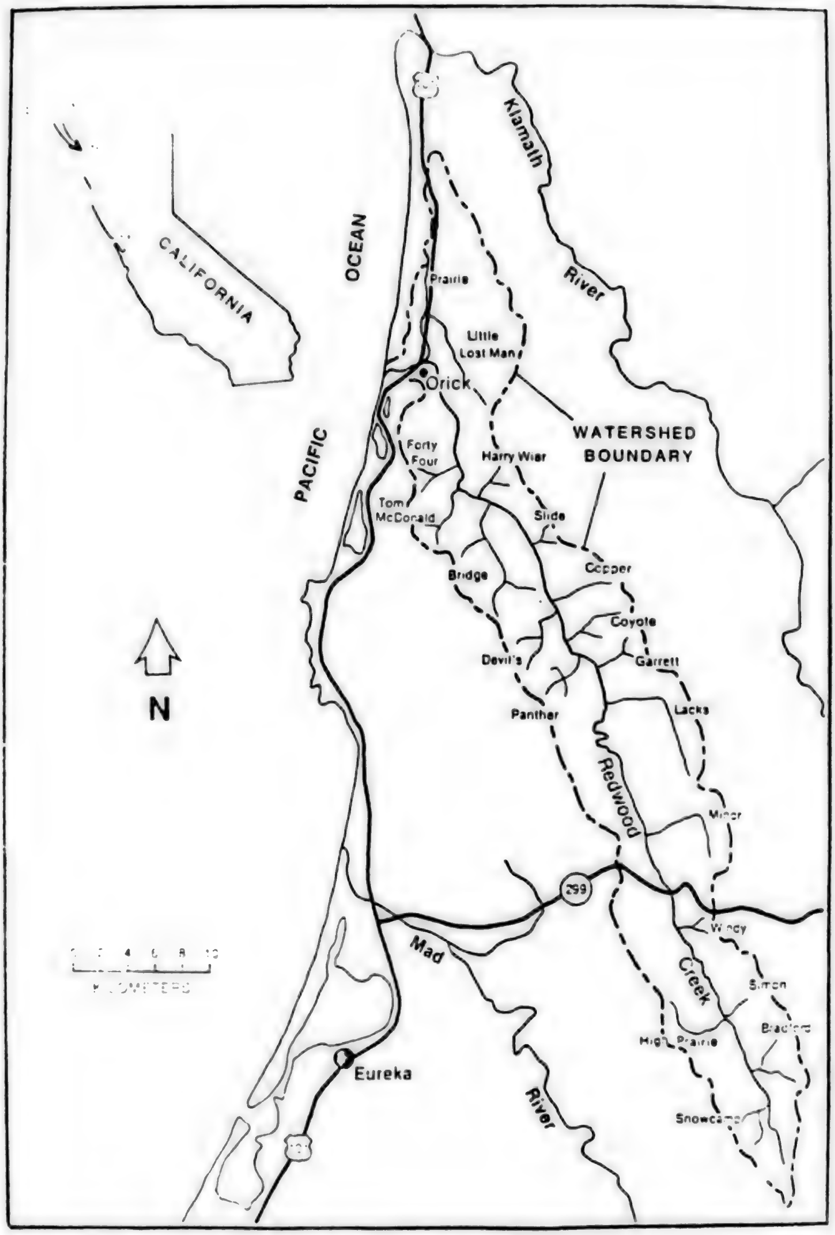

FIGURE 1. Redwood Creek, Humboldt County, California.

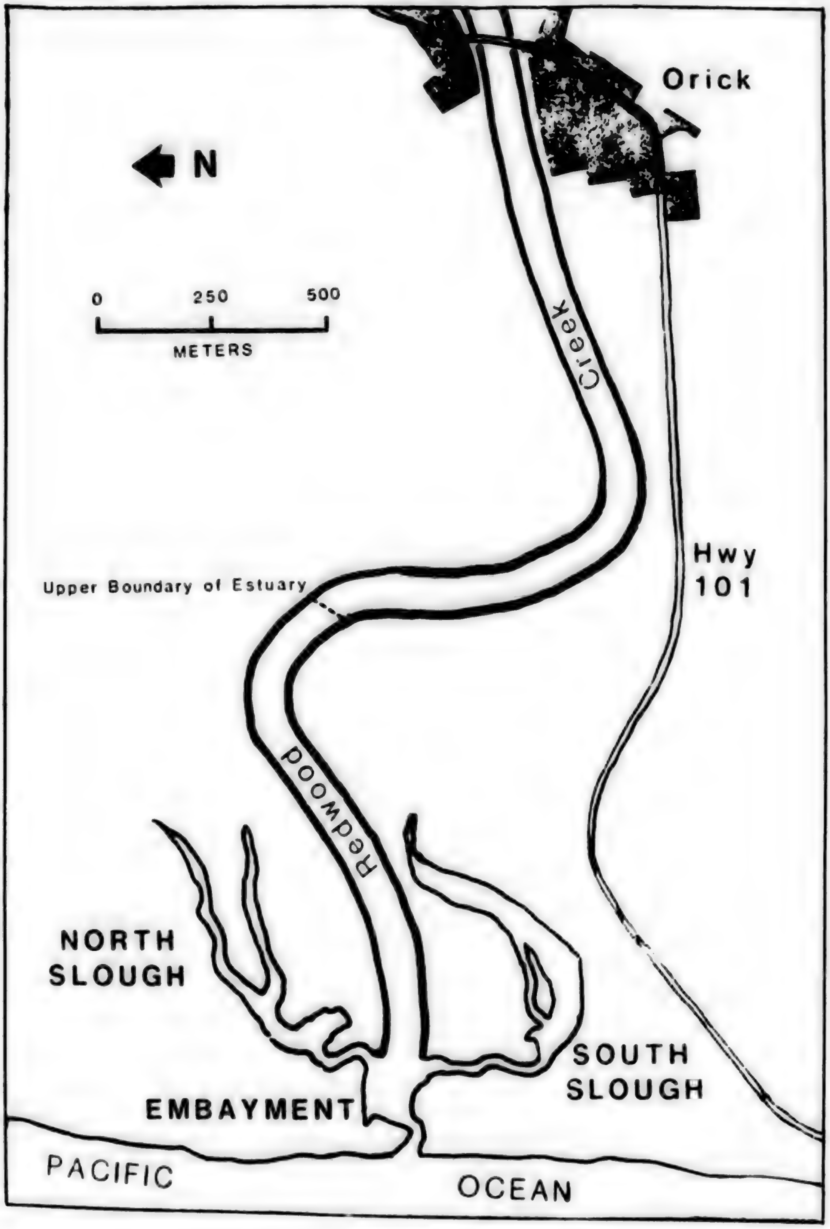

FIGURE 2. Lower Redwood Creek including the stream below the U.S. Highway 101 bridge, the estuary, and the North and South sloughs, Humboldt County, California.

We identified food items to the lowest possible taxon. Diet comparisons were made by using the Index of Relative Importance (IRI) (Pinkas et al., 1971):

$IRI = (\%N + \%W)(\%F)$

where N = numerical abundance of a prey item,

W = reconstructed dry weight of a prey item (computed by multiplying the number of individuals by the average dry weight of representative whole organisms), and

F = frequency of occurrence of a prey item.

Schoener's index (a) (Schoener 1970) was used to measure levels of diet overlap:

$a = 1 - 0.5 (\Sigma | p_x - p_y |)$

where Σ = the sum of the differences over all food categories,

p_x = the percentage, by reconstructed weight, of a particular food in the diet of species x, and

p_y = the percentage, by reconstructed weight, of the same particular food category in the diet of species y.

This index indicates that no food items were consumed in common if $a = 0.0$; and all food items were consumed in common and in equal proportions if $a = 1.0$. An $a \geq 0.60$ was used as the arbitrary boundary to identify biologically significant levels of diet overlap. This level of significance has been used by other investigators (Zaret and Rand, 1971; Mathur, 1977; Johnson, 1981a, 1981b; Johnson and Ringler, 1980; McCabe et al., 1983).

RESULTS

A total of 589 fish, representing 11 species, were collected for this study (Table 1). Sample sizes in some combinations of location and season were too small to allow meaningful comparisons of food habits. High stream flows precluded effective sampling during the winter and woody debris made sampling very difficult in the North Slough. The food habits of all species are presented in this report, but overlap comparisons are limited to situations where there was a sample size of at least four fish of each species for a given location and season. Insect larvae and particularly dipteran larvae and/or pupae were important in the diets of juvenile steelhead trout, juvenile chinook, juvenile coho, prickly sculpin, Sacramento sucker, and threespine stickleback in the area upstream from the estuary during all seasons (Table 2). In all cases, dipterans contributed more than 50% of the IRI except for juvenile coho, for which trichopteran larvae were most important in the spring; and prickly sculpin, for which ephemeropteran nymphs were most important in the summer.

Within the estuary, insects, primarily dipterans, dominated the spring and summer diets of juvenile steelhead, juvenile chinook, prickly sculpin, Humboldt sucker, and threespine stickleback (Table 3). Amphipods, *Corophium* spp., were most important in the fall diets of juvenile steelhead and prickly sculpin, and in the diets of starry flounder during all

TABLE 1. Number and size of fish examined for food habits from lower Redwood Creek, Humboldt County, California, February-December 1980.

	Number			Length (mm)	
Species	**Total Number**	**Number with identified food items**	**Percentage with identified food items**	**Mean**	**Range**
Steelhead trout (*Oncorhynchus mykiss*)	79	76	96.2	115.3	28-221(FL)1/
Chinook salmon (*O. tschawytsha*)	78	78	100.0	72.8	36-111 (FL)
Coho salmon (*O. kisutch*)	5	5	100.0	71.8	33-157 (FL)
Prickly sculpin (*Cottus asper*)	101	89	87.1	63.4	30-140 (TL)2/
Staghorn sculpin (*Leptocottus armatus*)	32	30	93.8	89.0	60-110 (TL)
Starry flounder (*Platichthys stellatus*)	43	35	81.4	25.3	73-254 (TL)
Sacramento sucker (*Catostomus occidentalis*)	41	35	85.4	93.5	44-200 (FL)
Threespine stickleback (*Gasterosteus aculeatus*)	197	182	92.4	44.8	7-77 (TL)
Surf smelt (*Hypomesus pretiosus*)	6	6	100.0	154.0	45-178 (FL)
Tidewater gobie (*Eucyclogobius newberryi*)	5	5	100.0	37.8	31-41 (TL)
Shiner surfperch (*Cymatogaster aggregata*)	2	0	0.0	39.5	39-40 (FL)

1/ FL = fork length.
2/ TL = total length.

TABLE 2. Percentage of the total IRI contributed by the principal prey items found in the stomachs of fish captured upstream from the estuary of Redwood Creek, Humboldt County, California, February-December 1980. (WI = winter, SP = spring, SU = summer, FA = fall).

	Juvenile Steelhead		Juvenile Chinook			Juvenile Coho	Prickly Sculpin			Sacramento Sucker			Threespine Stickleback		
Season...	SP	SU	WI	SP	SU	SP	WI	SP	SU	WI	SP	SU	SP	SU	FA
Food Item Sample size...	7	12	2	5	3	4	3	10	15	4	2	5	2	38	5
Acari					1.4									0.4	
Corophium spp.															1.0
Ephemeroptera nymphs		1.1	11.6	5.0	1.8	6.5		3.3		56.2					25.7
Plecoptera nymphs	25.8						32.0			3.3		1.8			
Hemiptera nymphs & adults	1.6					1.2									
Coleoptera larvae	2.0		0.8					2.1		0.8				2.0	
Coleoptera adults				2.6	1.2	0.7	2.1			2.9					
Trichoptera larvae		10.9	7.3	16.7			2.0	6.2	1.1						
Diptera larvae	42.5	46.1	82.7	34.6	29.1	50.7	57.3	55.1	42.2	92.3	93.8	95.4	21.3	64.7	62.9
Diptera pupae	23.7	26.6	0.3	37.3	51.8	33.7				0.8	5.3	2.8	56.6	7.1	36.0
Fish		1.0													
Eggs											0.8				
Terrestrials	0.8	0.7	3.4	6.1	9.1	12.7		35.2					15.4	0.2	

TABLE 3. Percentage of the total IRI contributed by the principal prey items found in the stomachs of fish captured from the estuary of Redwood Creek, Humboldt County, California, February-December 1980.
(WI = winter, SP = spring, SU = summer, FA = fall).

	Juvenile Steelhead			**Juvenile Chinook**		**Prickly Sculpin**			**Staghorn Sculpin**			**Starry Flounder**				**Sacramento Sucker**		**Threespine Stickleback**		**Surf-smelt**
Season...	SP	SU	FA	SP	SU	SP	SU	FA	SP	SU	FA	WI	SP	SU	FA	SP	SU	SP	SU	FA
Food Item Sample size...	6	36	7	10	53	12	23	9	2	9	9	2	5	14	12	2	14	27	28	6
Neanthes succinea						3.4			100.00		26.0			3.0	25.2			0.5		
Octopus rubescens										18.2										
Acari					0.6													1.5		
Copepods								3.4								35.9		4.4		61.6
Neomysis mercedis			0.5							6.0	6.0		8.2					1.2		1.1
Gnorimosphaeroma oregonensis			1.8			2.1				1.0		0.9	10.8	4.2				0.3		
Corophium spp.	12.3	25.1	89.6			4.6	5.3	80.5		60.4	10.7	8.8	58.8	66.3	74.0			17.4		3.0
Anisogammarus confervicolus			1.1			0.7				1.8	57.3	82.4	4.5		0.4					32.6
Crangon franciscorum									0.2				6.7							
Emerita analoga	22.5																			
Ephemeroptera nymphs	3.4																			
Plecoptera nymphs						1.1						8.0								
Coleoptera larvae						2.1														
Trichoptera larvae	8.4			2.2		1.2														
Diptera larvae	39.3	49.3	1.1	20.6	40.0	84.1	85.6	5.5		8.0			14.1	19.4		64.1	98.9	68.1	98.7	
Diptera pupae	8.7	19.3	1.4	56.9	57.6					1.0							0.8	6.6	0.5	
Fish		4.3	2.7	3.4			1.1	9.8		0.9									0.1	0.5
Eggs							6.6			0.2			3.7							
Terrestrials	1.8	0.4	1.6	15.2	1.4	0.2		0.1		1.5									0.1	0.1

TABLE 4. Percentage of the total IRI contributed by the principal prey items found in the stomachs of fish captured from the South Slough of Redwood Creek, Humboldt County, California, February-December 1980. (WI = winter, SP = spring, SU = summer, FA = fall).

	Juvenile Steelhead	**Juvenile Chinook**	**Prickly Sculpin**		**Staghorn Sculpin**			**Starry Flounder**	**Sacramento Sucker**	**Threespine Stickleback**			
Season...	SU	SU	WI	SU	WI	SP	SU	WI	SU	WI	SP	SU	F
Food Item Sample size...	4	5	8	6	5	2	3	2	8	12	16	18	10
Acari												0.1	
Cladocerans		0.5									1.8	1.6	0.6
Ostracods			2.6								0.9		0.1
Copepods				2.2							3.3	10.2	
Neomysis mercedis	0.6	8.9		7.2	26.6	84.0	75.7			46.2	25.8	11.1	91.0
Gnorimosphaeroma oregonensis	1.4	4.0	46.2	82.1	44.3		1.0	63.6	65.3	2.4	8.5	13.1	0.4
Corophium spp.	18.0	53.5	18.0	0.9	15.8	10.5	23.0		19.9	26.1	4.5	46.5	0.5
Anisogammarus confervicolus			29.3					33.7		7.1			
Ischnura cervula naiads		5.5							1.0			1.1	
Plecoptera nymphs		0.9											
Corixid nymphs & adults	54.1										0.4	0.8	
Notonecta shooteri nymphs	2.7								3.0				
Glyphopsyche sp. larvae	1.6	0.7							2.1				
Diptera larvae		1.5	4.0	6.8	2.6	5.5		2.7	7.8	4.2	40.5	6.3	2.1
Diptera pupae	5.1	12.3					0.3			13.9	14.3	3.9	5.3
Fish		4.1										0.5	
Eggs												4.6	
Terrestrials	13.6	4.9											

seasons. Since juvenile chinook left the estuarine embayment when the berm was breached in early July, no information was available to characterize their food habits during the late summer or early fall. Staghorn sculpin fed exclusively on polychaete worms in the spring, predominately on *Corophium* spp. in the summer, and on the amphipod *Anisogammarus* in the fall. Copepods were the primary food of surf smelt in the fall. Although no positive identification was made of the stomach contents of the shiner perch, they appeared to have fed exclusively on algal material.

The South Slough was a relatively stable habitat throughout the study period. The food habits of fish captured in this area varied more than at other sites (Table 4).

Few fish were caught in the North Slough. Threespine stickleback fed primarily on copepods and dipteran larvae and pupae. Tidewater gobies fed primarily on dipteran larvae.

There were only four instances of biologically significant overlap between species and within sites and seasons. The threespine stickleback showed a high degree of overlap with both prickly sculpin and Sacramento sucker in the estuary during the summer. Chironomid larvae was the common food item in both cases. In the estuary during the fall, prickly sculpin and starry flounder both commonly utilized amphipods and isopods such that their diets significantly overlapped. The diets of prickly sculpin and staghorn sculpin in the South Slough significantly overlapped during the winter because both were feeding heavily on *Gnorimosphaeroma*.

DISCUSSION

Most fish in lower Redwood Creek fed upon autochthonous food items. Other investigators have reported similar observations for other Pacific coast estuaries (McCabe et al., 1983; Simenstad, 1983). Except in the sloughs, riparian vegetation within the study area had been largely removed by the construction of the levees, limiting contributions of food from terrestrial sources. Other than juvenile chinook salmon that fed primarily on drift organisms, the fish fed primarily on benthic organisms. McCabe et al. (1983) reported that subyearling chinook salmon in the lower intertidal area of the Columbia River estuary also fed primarily on drift in the form of dipteran adults.

The food habits of the fish did vary between habitats and seasons. McCabe et al. (1983) also reported seasonal and spatial differences in the food habits of fishes captured in the Columbia River estuary. The general characteristics of the food habits of fishes in the lower Redwood Creek habitats would suggest the potential for interspecific competition. However, when food items were precisely identified, there were only four indications of biologically significant dietary overlap between species within a location at a given season. We suspected that food was not a limiting factor in the Redwood Creek estuary during the period of this study because either there were relatively few fish occupying the habitat or there was an abundance of food items. It should be noted that the catastrophic embayment draining was an unusual event. We feel that if the berm had not been breached and the downstream migrating salmonids were concentrated in the embayment throughout the summer, preferred food items such as chironomid larvae or *Corophium* could become limiting and affect fish production, as was observed by Nielson et al. (1985) in the Sixes River.

The construction of the levees has eliminated riparian habitat and cover in the form of cutbanks in the lower reaches of Redwood Creek. Also, the resulting channelization has altered circulation patterns, causing sedimentation within the estuary (Ricks, 1985). The National Park Service policy is to restore the estuary and to enhance its general productivity and, in particular, its productivity with regard to salmonids (Hofstra, 1983). Since the removal of the levees is not practical, other means of increasing the productivity of the area must be considered. A stable estuarine habitat during the summer should enhance the production of the invertebrate fauna upon which the fish depend. Population and growth data have verified that the food resources are adequate to ensure the growth of juvenile salmonids in the estuary through the summer and early fall (Larson et al., 1983). Redwood National Park is presently managing the embayment water levels in the summer and fall. Also, a diversion control structure has been constructed to restore circulation of water through the South Slough to improve water quality and accessibility for fish. In our opinion, these two actions are the most appropriate efforts to maintain and improve the productivity of the Redwood Creek estuary.

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Visitor Crowding and Conflict at Whiskeytown: A Social Carry Capacity Approach[1]

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Abstract. Social carrying capacity was used as the conceptual framework for looking at visitor crowding and conflict at Whiskeytown Unit of the Whiskeytown-Shasta-Trinity National Recreation Area. Close to 3200 visitors were contacted during the summer of 1985 at 16 selected locations around the lake. Findings focus on visitor perceptions of crowding at various locations at Whiskeytown and conflicts between visitor groups. Potential social carrying capacity problem areas and conflicting recreation activities are noted and discussed. Suggestions are presented for establishing a long-term visitor monitoring system based on the carrying capacity framework.

INTRODUCTION

In 1984 the National Park Service and Oregon State University began a research project at the Whiskeytown Unit of the Whiskeytown-Shasta-Trinity National Recreation Area. The objectives of the study were to provide Unit managers with a description of current use patterns on the lake and describe areas of conflict among user groups and between recreation activities.

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[1] A longer version of this paper will be published as a CPSU Report by the Cooperative Park Studies Unit at Oregon State University as part of Cooperative Agreement CA-9000-3-003, Sub-agreement No. 5.

Whiskeytown National Recreation Area (NRA), located near Redding in northwest California, is within a 3-hour drive of several million people. Whiskeytown Lake, a 1300-hectare reservoir created as part of California's Central Valley Project, is maintained at a constant level in summer and is surrounded by high slopes and extensive backcountry. The climate is moderate, with cool winters and hot, dry summers. Whiskeytown is popular with a variety of user groups who participate in a wide range of activities. Swimming, fishing, sunbathing, motor-boating, water skiing, sailing, picnicking, camping, and gold panning are the most popular activities.

Visitors to Whiskeytown tend to concentrate at locations particularly suited for recreation activities such as developed beach areas, coves, and secluded shoreline beach areas. Consequently, recreation activities may overlap in space and time, such as sailboat and motorboat users using the lake at the same time, and swimmers and anglers sharing secluded coves. Whiskeytown managers were concerned that this situation might result in competition for space among visitors, resulting in crowding and/or conflicts between user groups.

One strategy for looking at crowding among recreationists is to use carrying capacity, particularly the notion of social carrying capacity (Shelby and Heberlein, 1986). In many outdoor recreation situations, social carrying capacity is the most limiting factor on visitor use levels. Physical capacities in natural areas are often high, and social norms often restrict the acceptable number of people that can use an area before the physical capacity is met.

The purpose of this study was to use social carrying capacity as a conceptual framework to assess crowding and conflict at Whiskeytown. Potential social carrying capacity problem areas and conflicting recreation activities were identified, and alternative management processes to monitor visitor use over time are presented. Specific objectives included:

1. Establish baseline parameters for assessing social carrying capacity at Whiskeytown based on visitor perceptions of crowding;
2. Provide managers with descriptions of conflicts between user groups; and
3. Outline established processes for monitoring visitor use over time.

BACKGROUND

The long-established wildlife and range management concept of carrying capacity, the number of animals in a given habitat (Dasmann, 1964), began to receive renewed attention by recreation researchers in the 1950s and 1960s because of rapidly expanding recreation participation patterns.

Wagar (1964) suggested that increasing numbers of visitors affect the quality of the recreation experience as well as impact the environmental resources in an area. Together these impacts affect visitor satisfaction depending on the visitor's needs and motivations.

Lucas (1964) found that perceptions of crowding varied between different user groups. In his study of the recreation carrying capacity of the Boundary Waters Canoe Area in Minnesota, Lucas found that sensitivity to crowding differed in comparisons among paddlers, motor canoeists and motor boater users. Lucas concluded that a range of carrying capacities was needed based on different crowding values.

The role of management activities in affecting carrying capacity first received consideration by Wagar (1968). He expanded the carrying capacity concept beyond environmental and social concerns to include a management component that was appropriate in situations where the quality of recreation experiences might be enhanced by management actions. As examples, Wagar cited fertilization and irrigation practices to enhance vegetation, and the periodic rest and rotation of impacted sites. Using a similar strategy, the quality of recreation experiences might be improved by redistributing visitors to lesser used areas or by educating visitors to encourage more desirable visitor behavior.

Current formulations of carrying capacity have built upon this three-dimensional framework of environmental, social, and management factors. Shelby and Heberlein (1986) identified four types of carrying capacity and recognized that any one or a combination of these four parameters may limit the appropriate use level of a resource: ecological capacity, physical capacity, facility capacity, and social carrying capacity.

Social Carrying Capacity and Perceived Crowding

Social carrying capacity, the focus of this study, is often more difficult to assess than the other three types because it deals with impacts on the visitor's recreation experience. Social carrying capacity is based on human values and attitudes which cannot be directly manipulated by managers. Perceived crowding is one way visitors evaluate social capacities (Shelby and Colvin, 1981).

"Crowding" is a subjective term used to define a specific density or number of people in a specified area. An individual's psychological determination of a crowd is based on his/her definition of the appropriate number of people in a certain situation (Shelby and Heberlein, 1986). For example, 500 people at a concert in a park may not be defined as crowded, but 500 people around a small alpine lake probably would be. Depending on the normative definition of the situation, people can feel crowded regardless of the amount of physical space available.

Shelby and Heberlein (1986) suggested that one way perceived crowding can be used to approximate an evaluative standard for appropriate levels of use as an indirect means of assessing social carrying capacity is to assess the percentages of visitors who feel crowded. Comparing data from 22 different studies that utilized the same single measure of crowding, the authors propose a "one-third two-thirds" rule, suggesting that social carrying capacity has probably been exceeded if more than two-thirds of visitors feel crowded. If fewer than one-third feel crowded, the area is probably below carrying capacity. When perceived crowding falls between these two thresholds, no determination can be made using this standard.

Conflict

The second objective of the study was to provide Whiskeytown managers with descriptions of conflicts between visitor groups. There are many causes for conflicts among recreationists using limited resources. Recent research suggests that a key contributor to conflict among recreationists is goal interference: conflict results when the behavior of one group of recreationists is incompatible with the physical, social, or psychological goals of another group (Jacob and Schreyer, 1980; Gramman and Burdge, 1981; Jackson and Wong, 1982).

Gramann and Burdge (1981) studied conflicts between water skiers and fishermen and determined the problem was due to incompatibility of recreation goals between the two groups. Fishermen who reported tranquility, nature enjoyment, and tension release as important reasons for fishing were more bothered by water skiers than fishermen whose goals were thrill-seeking, exercise, or meeting and observing people. Driver and Bassett (1975) attributed conflict among canoeists and fishermen on the Au Sable River to incompatibilities in the kinds of psychological satisfactions desired from an experience, as well as differences in the groups' definitions of appropriate social behavior and contact levels. River users defined unacceptable behavior and a lack of consideration as the primary conflicts.

Studies of whitewater river users describe conflicts between motorized and non-motorized travelers (Schreyer et al., 1976; Shelby and Nielson, 1976). Floaters on non-motorized trips generally felt motorized use was inappropriate for such areas. However, motorized users were often not at all bothered by non-motorized users. Thus, conflicts between user groups may often be one-sided, contributing to difficult carrying capacity decisions for recreation resource managers.

As with perceived crowding, conflicts tend to increase as contact levels increase. However, contact levels are not always directly related to use levels (Heberlein and Vaske, 1977), so managers should not attempt to alleviate conflict simply by decreasing use. What is important is to identify "the conflict potential of various recreation resource clientele ..." (Jacob and Schreyer, 1980) and determine the appropriate ranges of social carrying capacities.

APPLYING THE CARRYING CAPACITY CONCEPT

The third objective of the study was to outline alternative management planning processes for monitoring visitor use over time. Three models have emerged from the resource management literature which apply research findings to management issues related to carrying capacities: Limits of Acceptable Change (LAC), Carrying Capacity Assessment (C-CAP), and Visitor Impact Management (VIM).

Limits of Acceptable Change (LAC). One of the first applications of carrying capacity research to recreation resource management was the model of LAC applied to wilderness settings (Stankey, 1973). The LAC model has more recently been formalized as a nine-step planning process (see Stankey and McCool, 1984). The focus of the LAC process is on "management of the environmental and social conditions identified as desired" (p. 466). Rather than focusing on use levels, the emphasis is on the location, amount, and type of change considered appropriate and acceptable as set forth in the management objectives set forth for a particular area.

Carrying Capacity Assessment Process (C-CAP). Shelby and Heberlein (1986) developed a detailed assessment process for developing carrying capacity studies and monitoring impacts. The process, as with LAC, focuses on acceptable levels of change. It incorporates use of visitor-defined encounter norms in developing standards for evaluating impacts.

Visitor Impact Management (VIM). Graefe et al. (1986) provided a discussion of factors affecting types of impacts, principles for managers to use in evaluating impacts, and a framework to guide decision-making in visitor impact management. The authors contend that prior to determining tolerable levels of impact, recreation resource managers must understand the nature of those impacts whether they be physical, ecological, or social. Incorporating consideration of factors influencing impacts, the authors suggest a sequential process aimed at reducing or controlling impacts that threaten "the quality of outdoor recreation areas or experiences" (p. 433).

The carrying capacity models described above have been applied in several different cases. Each provides a useful process for developing strategies to monitor impacts associated with recreational use and are presented to Whiskeytown managers as potential visitor monitoring strategies.

METHODS

Data were gathered from 16 sites around Whiskeytown Lake (see Figure 1). Sites were chosen to reflect the variety of physical settings, recreation activities, and visitor use levels around the lake. From May 26, 1985 to September 2, 1985, approximately 200 surveys were distributed at each site during systematically sampled time blocks stratified by weekdays, weekend days, and holidays.

The primary data collection instrument was a mailback questionnaire distributed on-site to visitors 16 years and older by trained interviewers. Two follow-up reminders were sent, with a total of 2313 usable questionnaires being returned which resulted in a 75% response rate. Perceived crowding was assessed using the 9-point crowding scale developed by Shelby and associates and comparisons were made among visitors at the 16 interview locations. Visitors were asked to indicate how crowded they felt at that particular location, with possible responses ranging from "not at all crowded" to "extremely crowded." Visitors were also asked to estimate the number of people they could see at one time. Social carrying capacities for the 16 locations were compared using the "one-third two-thirds" rule (Shelby and Heberlein, 1986).

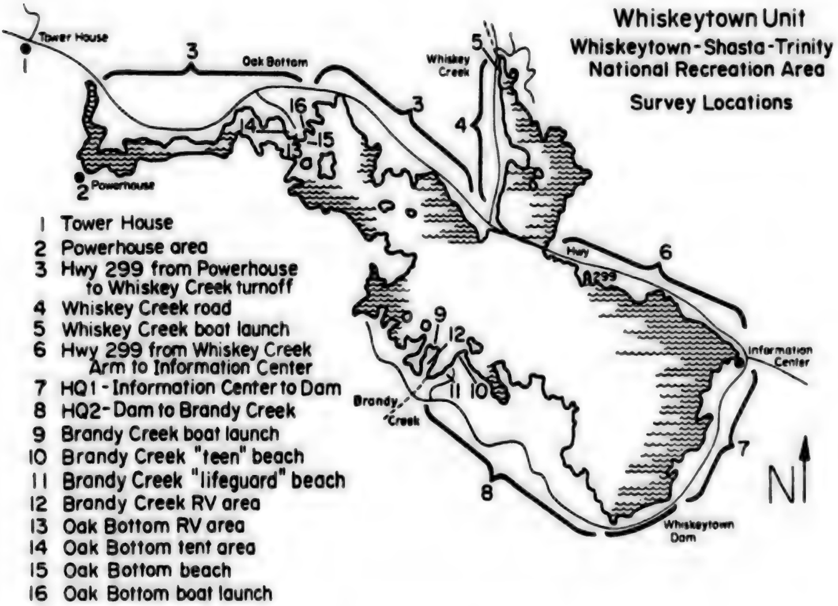
Whiskeytown Unit
Whiskeytown-Shasta-Trinity National Recreation Area
Survey Locations
Tower House
Oak Bottom
Whiskey Creek
Powerhouse
Hwy 299
Information Center
Brandy Creek
Whiskeytown Dam
N
1 Tower House
2 Powerhouse area
3 Hwy 299 from Powerhouse to Whiskey Creek turnoff
4 Whiskey Creek road
5 Whiskey Creek boat launch
6 Hwy 299 from Whiskey Creek Arm to Information Center
7 HQ1 - Information Center to Dam
8 HQ2 - Dam to Brandy Creek
9 Brandy Creek boat launch
10 Brandy Creek "teen" beach
11 Brandy Creek "lifeguard" beach
12 Brandy Creek RV area
13 Oak Bottom RV area
14 Oak Bottom tent area
15 Oak Bottom beach
16 Oak Bottom boat launch

FIGURE 1. Sixteen survey locations, Whiskeytown, Summer 1985.

Conflicts were assessed by asking visitors whether any group had interfered with their activities at Whiskeytown. Those who responded "yes" were asked to indicate their location at the time of interference, what activity they were involved in, and the activity of the interfering group(s). From these data, areas where conflicts most often occurred and conflicting activities were identified.

RESULTS AND DISCUSSION

This study is representative of visitors at the 16 Whiskeytown sampling locations who responded to the survey. Given the high response rate, these data probably reflect all visitors contacted, including those who failed to return the questionnaire. Whether or not these visitors adequately represent all visitors to Whiskeytown cannot be determined from available data. While this limitation must be recognized, it should also be noted that sampling occurred at the majority of locations frequented by visitors to Whiskeytown.

Crowding

Based on mean responses to the crowding scale, ranging from 1 = not at all crowded to 9 = extremely crowded, visitor responses for the 16 interview locations are compared in Table 1. Significant differences among mean values were tested using the Student-Newman-Keuls multiple range test (Steel and Torrie, 1980). Estimates of the number of other people visitors saw at each location are presented in Table 2.

Visitors at the Oak Bottom tent camping area felt most crowded, followed by visitors at the Brandy Creek "lifeguard" and "teen" beaches. The two Brandy Creek beaches and the Oak Bottom beach are popular

TABLE 1. Crowding means by location.

	Mean[a]	n	Differences among means[b]						
Oak Bottom tent	5.0	154	A						
"Lifeguard" beach	4.5	151	A	B					
"Teen" beach	4.5	135	A	B					
Oak Botton RV	4.0	142		B	C				
Oak Bottom beach	3.8	145			C				
HQ1 shoreline	3.6	135			C	D			
Oak Bottom boat launch	3.5	150			C	D			
Whiskey Creek boat launch	3.4	147			C	D	E		
Whiskey Creek road shoreline	3.0	133				D	E		
Brandy Creek boat launch	2.9	146				D	E	F	
Brandy Creek RV	2.7	152					E	F	
Powerhouse	2.6	153						F	
Hwy 299-Powerhouse to WC shoreline	2.5	133						F	
HQ2 shoreline	2.5	126						F	
Hwy 299-WC to Info shoreline	2.3	142						F	
Tower House	1.6	126							G

[a] Mean of crowding scale where 1 = not at all crowded ... 9 = extremely crowded.

[b] Means with the same letter are not significantly different at the .05 level using Student-Newman-Keuls test.

TABLE 2. Number of other people seen by location.

Location	< 20 n(%)	21-50 n(%)	51-100 n(%)	101+ n(%)	Couldn't estimate n(%)
Towerhouse	119(92)	9(7)		1(.8)	
Powerhouse	134(86)	16(10)	3(2)		1(.6)
Hwy 299-PH-WC shoreline	121(91)	10(7)	2(2)		
Whiskey Creek road to shoreline	92(69)	34(26)	4(3)	2(2)	1(.8)
Whiskey Creek WC boat launch	104(70)	33(22)	6(4)	1(.7)	5(3)
Hwy 299-WC-Info. shoreline	135(95)	5(3)	2(1)		
HQ1 shoreline	81(60)	41(30)	9(7)	1(.7)	3(2)
HQ2 shoreline	112(90)	10(8)			2(2)
Brandy Creek boat launch	66(46)	48(33)	18(12)	11(8)	1(.7)
Brandy Creek "Teen" beach	14(11)	28(21)	35(26)	50(38)	6(4)
Brandy Creek "Lifeguard" beach	16(11)	28(19)	52(35)	44(29)	10(7)
Brandy Creek RV area	111(72)	27(18)	7(5)	6(4)	2(1)
Oak Bottom RV area	46(32)	49(34)	34(24)	10(7)	5(3)
Oak Bottom tent area	91(60)	42(27)	9(6)	5(3)	6(4)
Oak Bottom beach	21(15)	31(22)	59(41)	28(20)	3(2)
Oak Bottom boat launch	72(49)	50(34)	14(10)	7(5)	3(2)

sites at Whiskeytown. It is not unexpected that visitors at these locations reported seeing more people than visitors at other sites (e.g., 38% of "teen" beach visitors reported seeing more than 100 people at one time). All three areas are fairly open with many opportunities for seeing, hearing, and interacting with other visitors.

Although the Oak Bottom tent campers reported seeing fewer numbers of people (60% saw 20 or fewer), they felt most crowded. Research has shown that contacts with others at campsites are evaluated more negatively than contacts elsewhere (Stankey, 1973). People expect more privacy at campsites, which may be viewed as their temporary residences. This may in part explain why the Oak Bottom te[illegible]rs felt crowded even though they saw relatively fewer other people [illegible] the area. Many campsites offer little screening from other visitors and during the summer months the campground is often filled to capacity. A somewhat similar situation exists at the Oak Bottom RV area, where no screening is available and a great deal of foot and vehicle traffic occur.

A mean score between 3.0 and 4.0 indicated visitors felt slightly crowded. This rating was expressed by respondents contacted at the three boat launch sites, the HQ1 shoreline area, the Whiskey Creek road shoreline, and the Brandy Creek RV area. Crowding at boat launch areas is not unexpected, particularly at Oak Bottom where there is congestion from beach users and overnight campers in addition to boat launching traffic, all likely contributing to perceived crowding. Visitors at the remaining shoreline locations, including the Powerhouse areas and particularly those at the Tower House historic site, felt the least crowded.

Crowding is a complex phenomenon influenced by many factors. It is unlikely that perceived crowding is, by itself, a valid criterion for determining social carrying capacity. However, perceived crowding can be used as a rough indicator for identifying carrying capacity problem areas and making comparisons between areas. The 16 Whiskeytown sites were grouped according to the percentages of respondents who felt crowded; these groupings are presented in Figure 2. According to the "one-third two-thirds" rule, four areas may have exceeded social carrying capacity: the Oak Bottom tent camping area, the Brandy Creek "lifeguard" beach, the Brandy Creek "teen" beach, and with 67% of respondents feeling crowded, perhaps the Oak Bottom beach. The two areas likely below social carrying capacity, according to this criterion, are the Highway 299 shoreline (between Whiskey Creek and the Information Center) and the Tower House. The bulk of the Whiskeytown sampling locations fall in the middle, where judgments on social carrying capacity are uncertain.

It was interesting to find that the HQ1 shoreline visitors felt noticeably more crowded than the visitors at the other four shoreline areas. The heavily used HQ1 shoreline area appears to lack opportunities for solitude and privacy that may draw visitors to shoreline areas.

Visitor Conflicts

Conflict between visitor groups was a concern to Whiskeytown managers. Conflict can occur between groups pursuing different activities because of, among other things, differences in motivations, goals, and the perceived inappropriate behaviors of other groups.

Twenty percent of survey respondents reported that another group had interfered with their activities at Whiskeytown. Visitors who felt interference tended to be swimming (21%), fishing (17%), sunbathing (11%), resting/relaxing (10%), sailing (9%), and doing campsite activities such as cooking and eating (8%). A few were motor-boating (6%) and waterskiing (4%), while the remainder (14%) listed miscellaneous other activities.

The interview locations of the six most frequently-mentioned activities of those who reported interference are presented in Table 3. The question about interference dealt with a respondent's entire Whiskeytown visit and it was assumed that the probability of the interference occurring at the interview location was fairly high, although it could have occurred elsewhere. Swimmers reporting interference were found primarily at shoreline locations, particularly HQ1 and HQ2. Anglers were contacted most often in the powerhouse area, the Whiskey Creek boat launch, and along shoreline areas. Sunbathers were found primarily on shorelines and the Brandy Creek "teen" beach. Those who were resting and relaxing when they were interfered with were most often at the Oak Bottom tent area, Oak Bottom boat launch, and Brandy Creek RV area. Visitors interfered with while sailing were contacted at the Brandy Creek and Oak Bottom boat launches and those doing campsite activities at the three campsite locations.

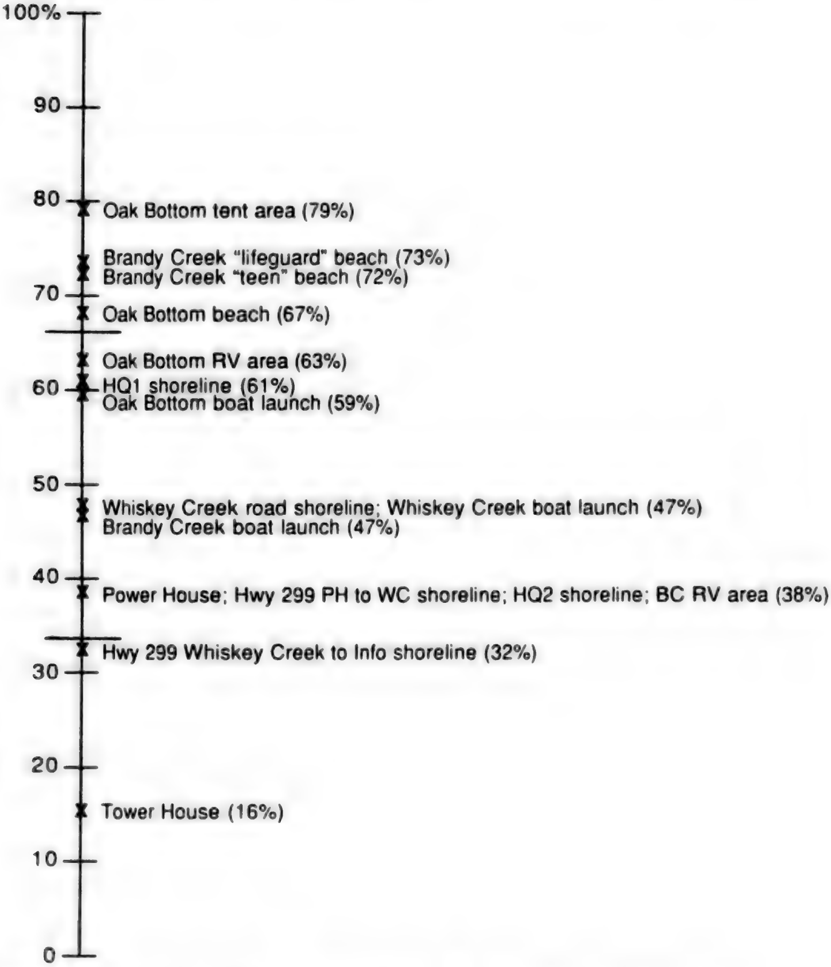

FIGURE 2. Percentages of Whiskeytown visitors who felt crowded, by location.

TABLE 3. Locations[a] of six most frequent activities of respondents who felt interference from others.

Swimming (n = 90)

- HQ2 shoreline (20%)
- HG1 shoreline (17%)
- Whiskey Creek road shoreline (12%)
- Hwy 299-WC to Info. shoreline (12%)
- Hwy 299-PH to WC shoreline (6%)
- Brandy Creek "Teen" beach (6%)
- Powerhouse (4%)
- Brandy Creek boat launch (4%)
- Oak Bottom beach (4%)
- Others (15%)

Fishing (n = 76)

- Powerhouse (25%)
- WC boat launch (16%)
- HQ2 shoreline (9%)
- Oak Bottom RV (9%)
- Hwy 299-PH to WC shoreline (8%)
- Hwy 299-WC to Info. shoreline (5%)
- Oak Bottom boat launch (5%)
- HQ1 shoreline (5%)
- Others (18%)

Sunbathing (n = 48)

- Hwy 299-WC to Info. shoreline (17%)
- Brandy Creek "Teen" beach (15%)
- HQ1 shoreline (12%)
- Hwy 299-PH to WC shoreline (12%)
- WC road shoreline (10%)
- Powerhouse (6%)
- HQ2 shoreline (6%)
- Brandy Creek "Lifeguard" beach (6%)
- Others (16%)

Resting/Relaxing (n = 43)

- Oak Bottom tent (23%)
- Oak Bottom boat launch (14%)
- Brandy Creek RV (12%)
- Tower House (7%)
- HQ1 shoreline (7%)
- HQ2 shoreline (7%)
- Oak Bottom RV (7%)
- Others (23%)

Sailing (n = 41)

- Brandy Creek boat launch (63%)
- Oak Bottom boat launch (22%)
- Others (15%)

Campsite activities (n = 35)

- Oak bottom tent area (37%)
- Oak bottom RV area (15%)
- Oak bottom beach (9%)
- Brandy Creek RV area (9%)
- Others (30%)

[a] Interface did not necessarily occur at the interview location. Question dealt with respondent's entire Whiskeytown visit.

Table 4 presents activities that conflicted with the most frequently mentioned activities of those reporting interference. The most notable finding is that motorboating and, to a lesser extent, water skiing, conflicted with virtually all activities mentioned. Swimmers and anglers particularly reported interference from motorboats. Interferences including speeding, coming too close to shore, making large wakes, disturbing fish, invading privacy, and noise. Anglers also reported having problems with swimmers

TABLE 4. Conflicting activities.

Activities of those who felt interference[a]	Activities of those who interfered[b]								
	Motorboating	Water Skiing	Swimming	Walking	Pets/ Animals	Loud Music	Yelling	Campsite Activities	n
Swimming	45(50%)	19(21%)							90
Fishing	36(47%)	20(26%)	11(14%)						76
Sunbathing	12(25%)	4(8%)		5(10%)		5(10%)			48
Resting/relaxing	8(19%)					14(33%)			43
Sailing	28(68%)	5(12%)	4(10%)						41
Campsite activities						13(37%)	8(23%)	6(17%)	35

[a] Six most often mentioned activities.

[b] Most often mentioned activities.

getting in the way while fishing and launching and loading boats. Sunbathers were also bothered by motorboats and also listed people walking through the area and allowing pets and animals running loose as interfering activities.

Many of the swimmers, sunbathers, and anglers who complained about motorboats and water skiers were contacted at shoreline locations. Often these areas do not have the restrictions on boat speed and wakes found at designated swimming areas and boat launch sites. Boaters and skiers may not realize that their wakes and noise are disturbing shoreline visitors and might, in some cases, be dangerous for swimmers.

Those who reported interference while sailing also most often cited motorboats as being the cause. Water skiers and swimmers were also a problem for those sailing, though to a much lesser degree.

Visitors reporting interference while resting and relaxing, the majority contacted at the tent and RV campground areas, most often mentioned playing loud music as the interfering activity, as did visitors engaged in campsite activities. Overnight visitors also complained of people yelling and other unacceptable campsite behaviors.

While there does appear to be some conflict between various user groups at Whiskeytown, it must be remembered that only a relatively small number of respondents (one-fifth of those surveyed) felt others interfered with their visit. It is important, however, to note and monitor those activities that appear to be in conflict and attempt to avoid problems if possible, particularly when potentially dangerous encounters between motorboats and/or water skiers and swimmers occur.

CONCLUSIONS

The results presented here have focused on crowding, social carrying capacity, and visitor conflicts at Whiskeytown. Based on these results, we present the following recommendations to managers. Some recommendations suggest management strategies for specific Whiskeytown locations, although much of what we recommend could be of use to resource managers concerned with visitor crowding and conflict issues.

Crowding

Although perceived crowding cannot be used as the sole determinant of social carrying capacity, it is useful for identifying potential over-capacity and problem areas. Several sites that merit monitoring emerged from an examination of visitor perceptions of crowding using the "one-third two-thirds" rule.

The Oak Bottom tent area was consistently perceived to be the most crowded area at Whiskeytown. Although the visitors at this site did not report seeing large numbers of visitors, the fact they saw them while in a campsite "home" may have contributed to their feeling crowded. The tent campground is a popular overnight area at Whiskeytown and is often full. Planting vegetation will provide a useful screen for separating camping groups and should be utilized as much as possible.

Past research has shown that people feel more crowded when they encounter more people than they expect and/or more than they prefer (Shelby et al., 1983; Ditton et al., 1983; Womble and Studebaker, 1981). Every effort should be made to make campers aware of the situation at the tent campground in order to minimize unrealistic expectations and perhaps decrease perceptions of crowding.

Educating visitors about what to expect may also be appropriate at the Brandy Creek "teen" and "lifeguard" beaches, two areas managers should view as having potential social carrying capacity problem areas. Visitors at both sites reported seeing large numbers of other visitors, to be expected given the open expanse of these heavily used beaches. Accurate information about the potential numbers and types of visitors encountered at each area would allow potential visitors to either adjust their expectations or select another setting more to their liking. Either reaction could reduce feelings of crowdedness among visitors at the beach areas. For example, informing visitors that the "lifeguard" beach is a good place for families with children, while the "teen" beach offers good sunbathing opportunities but has no lifeguard, will provide them with more accurate expectations to use in making choices about where to recreate.

The shoreline area with potential crowding and capacity problems is HQ1, the shoreline from the Information Center to the dam. Use is concentrated at a few heavily used sites along the shoreline and crowding may become a problem. The shoreline areas at Whiskeytown are unique in that they not only provide access to the lake, but also provide opportunities for privacy and solitude not available at the more developed beach areas. Some of the sites, such as HQ1, have toilets and garbage cans. These quasi-developed areas perhaps provide an experience that falls somewhere between a developed beach area and a primitive shoreline site.

Managers must consider the type of experience they want to provide at the HQ1 shoreline site and others like it. If opportunities for privacy are desired and no longer being provided, it may be necessary to consider limiting use at the HQ1 shoreline site. Other shoreline sites, though not currently experiencing crowding problems, should be viewed in the same light—what type of recreation opportunities do managers want to provide?

The Oak Bottom RV area is the other overnight area that may have a social carrying capacity problem. Given the lack of screening and congestion from boat trailers, it is not surprising these people feel crowded. In addition to making potential RV users aware of conditions at the RV campground, it might also help to inform them of the much less crowded Brandy Creek RV campground. A long-term partial solution to the crowding problem at the Oak Bottom RV area would be to remove a portion of the asphalt lot and plant trees and other screening vegetation.

Conflicts

Relatively few conflicts were found between groups at Whiskeytown, but those reported most often involved motorboats. Swimmers, sunbathers, and anglers at shoreline areas in particular noted problems from motorboats, such as excessive speed, being too close to shore, and wakes. Encounters between boats and swimmers can also be potentially dangerous. Speed limits and no wake signs are currently posted near designated swimming areas and it may be necessary to restrict boat speed near shorelines or prohibit boats and water skiers from coming too close to shoreline areas.

Boaters and skiers should be made aware of the problems caused by wakes and the potential danger in coming too close to shorelines where people might be swimming. In turn, those using shoreline areas should be made aware of possibilities and potential dangers of encounters with boats and skiers.

Areas most heavily used by anglers could be made off-limits to motor boats, water skiers, and swimmers. In areas where segregating uses is not feasible, boat speed limits should be instigated and enforced. Swimmers need to be made aware of fishing use in these areas to minimize conflict between these users.

Loud music interfered with visitors contacted primarily at the tent campground, who were resting or involved in campsite activities. People playing loud music and yelling (another activity mentioned) are common problems in campgrounds where sites are in close proximity. Rules regarding quiet hours should be posted and enforced and visitors urged to be considerate of others.

The common theme in the recommendations discussed above is to increase visitor awareness of potential conflicts between users. Research has shown that visitor groups often may not be aware that their actions are causing problems for other users. For example, motor-boaters may not be aware of the problems they cause by coming too close to shore and creating large wakes. Making them aware of the consequences of their actions, along with enforcement of speed limits, may decrease these conflicts. The same logic applies to campground users. Awareness of problems caused by inconsiderate behavior will hopefully decrease occurrences of conflicts between visitors.

Monitoring Impacts Over Time

Efforts have been made to describe a process for applying the carrying capacity concept to managing recreation resources. Three models were briefly described above–Limits of Acceptable Change (LAC); Carrying Capacity Assessment Process (C-CAP); and Visitor Impact Management (VIM). It was not the intent of this research effort to suggest one model over another. Rather, the similarities among the three models are discussed as important considerations for managers in setting up a visitor use monitoring system at Whiskeytown.

All three models focus on managing the environmental and social conditions identified as desired, including periodically monitoring existing conditions to ensure that the impacts fall within acceptable limits. All three include consideration of social, physical, and biological carrying capacities.

One of the first steps common to all three models is to identify the type of experience opportunities to be provided. For example, Whiskeytown managers might decide that the shorelines should be managed to provide opportunities for seclusion and privacy, and that the developed beach areas should be managed to provide opportunities for socializing and safe swimming for children.

Impact indicators, such as amount of vegetation lost or numbers of parties encountered, can be selected to reflect the kinds of conditions or opportunities managers are striving for. Standards can be specified for key impact indicators, such as "no more than 30% vegetation loss" or "encountering no more than 20 other parties per day." Existing site conditions can then be monitored and periodically compared to standards on the basis of baseline conditions. If any standards are exceeded, management strategies to conform site conditions to acceptable limits can be examined and implemented.

The park's management objectives can be used to answer the question of how much change is acceptable by deciding what type of recreation experience opportunities a particular setting should provide, including the level of naturalness, the kind of experience offered, and the intensity of management practices (Manning 1986). Numerous research studies emphasize the need for management objectives (e.g., Frissell and Stankey, 1982; Lucas and Stankey, 1974; Bury, 1976; Brown, 1979; Boteler, 1984).

Implementation of a carrying capacity monitoring system will involve not only an understanding of the concept, but also the development of a systematic approach to managing conditions. Regardless of the monitoring system adopted, key considerations include:

1. Review of management objectives;
2. Identification of the types of recreation experience opportunities to be provided;
3. Selection of key impact indicators that reflect desired opportunities;
4. Selection of standards for the key indicator variables;
5. Comparison of standards and existing conditions;

6. Development of management strategies to deal with impacts if needed;
7. Implementation of the selected strategy; and
8. Continuous monitoring of the impacts.

It is apparent that Whiskeytown offers a wide diversity of activities, facilities, social environments, and recreation experience opportunities. Managers can use information about this diversity along with management objectives in determining the types of experience opportunities to provide and the corresponding indicators and standards. Perceptions of crowding using the "one-third two-thirds" rule is an example of a social impact indicator and standard. Once the indicators and standards are developed, periodic monitoring of the conditions at Whiskeytown may reveal unacceptable changes in the environment or in the quality of visitor experiences over time that would otherwise go unnoticed.

This study provides managers with baseline data on existing and potential social carrying capacity, crowding, and conflict problems at Whiskeytown. Recommendations are based on these findings and, in addition, these data are offered as input into any of the three alternative visitor use monitoring systems discussed in the paper. Guided by management objectives, a monitoring system incorporating such data can help managers in their attempts to provide quality recreation opportunities to visitors while maintaining desired levels of environmental quality.

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SECTION II: FLORA

Introduction to the Section on Flora

One of the prerequisites for successful stewardship of national parks is detailed information on the natural resources that are being protected. This is best exemplified by the long-term ecological monitoring program at Channel Islands National Park (Davis, 1989). How else can the National Park Service, one of the leading conservation agencies in the world, evaluate its effectiveness in managing lands unimpaired for the enjoyment of future generations?

It is no accident that the papers presented at this conference focused on the techniques used in the inventory and monitoring of natural resources. The atmosphere at the conference was one of "urgency." This perceived urgency is due to increased impacts from a variety of sources, including (but not limited to) air pollution, habitat fragmentation, alien species invasions, and even the threat of global warming (see Schonewald-Cox and Stohlgren 1989a and b). At the same time, however, we realize that our environmental consciousness is rising faster than the funds appropriated for research. Thus, it is imperative to develop techniques to quickly, cheaply, and accurately assess the current status of our biotic and abiotic resources, so that the limited research dollars can be directed to the most important problems.

In their paper, Robinson et al. point out that the vascular plant inventories for 18 California units of the National Park Service are incomplete and lack uniformity. Plant lists, by their very nature, are generally incomplete due to a lack of past collections, misidentifications, and/or errors in nomenclature. Unless a concerted effort is made to continually update, authenticate and disseminate information in a timely fashion, the authors point out that it will be impossible to make comparisons among different conservation lands or between protected and unprotected lands. Moreover, there would be no way to assess the current status and distribution of plant diversity in these park lands, or to evaluate the parks' natural resource management programs. The current status of plant inventories in many parks is unflattering. Despite the valiant goals of the NPFLORA project (Waggoner and Thorwardson, 1983), the most current and complete listing of vascular plants for the 18 California national parks is the one compiled by Robinson et al. An important (albeit preliminary) finding of the authors is that about 65% of California's native vascular plants are found on NPS lands. The ongoing field inventories in Death Valley National Monument, Sequoia and Kings Canyon National Park, Yosemite National Park, and Pinnacles National Monument will likely increase that percentage.

Inventories of lichens also provide important information because these organisms serve as biomonitors for air pollution. Clifford W. Smith provides an annotated checklist of lichens, and their frequency and percentage cover in Pinnacles National Monument. These baseline data provide a means to assess long-term biological effects of increasing air pollution from the nearby San Jose metropolitan area. Only a handful of our national parks have detailed lichen surveys (USDI, 1988).

In addition to air pollution, alien plant invasions pose a significant threat to park ecosystems. Terri Thomas describes an alien plant inventory program at Golden Gate National Recreation Area (GGNRA) that can be emulated by national parks throughout the nation. She outlines a methodology that is logical and efficient, with four objectives: (1) to gather biological and environmental information necessary for setting priorities for controlling alien plant populations; (2) to evaluate eradication techniques; (3) to monitor changes in alien plant populations; and (4) to identify research needs. She draws on several examples from GGNRA which have analogs in many park ecosystems. While the species of invading plants will differ among parks, the approach need not.

John M. Randall complements Thomas's extensive alien plant inventory by intensively studying the invasions of bull thistle (*Cirsium vulgare*) in Yosemite Valley meadows. The invasibility of a plant community is often determined by complex interactions of disturbance, competition, microenvironmental attributes and the species' reproductive and dispersal traits (see Rejmanek, 1989 for review). Randall points out that the establishment and spread of bull thistle is heavily influenced by soil disturbance and soil moisture levels, as well as the cutting techniques commonly used to control this weed. His paper illustrates the type of sound research necessary to base resource management decisions on controlling "one of the park's three worst weeds."

The section concludes with another paper by Terri Thomas that quantifies changes in plant species composition following removal of cattle grazing in the Tennessee Valley area of GGNRA. Again, the battle lines are drawn between alien grasses and forbs and California's native perennial grasses (see Menke, 1989). Here grazing history, species composition, soil parent material, and annual fluctuations in vegetative cover combine to influence grassland succession. The recommendation to propagate and plant native grass plugs on a continual basis reminds us that restoring a "vignette of primitive America" may require our intervention for perpetuity. Thus, it is doubly important that we base management options on detailed inventory and monitoring data and sound research, for the parks of tomorrow will directly reflect the decisions we make today.

In summary, the five papers in this section are important because they contribute to our understanding of some park resources. They point out how little we know about most other park resources and the processes that shape them. Future research inventory and monitoring efforts must be based on service-wide standardized techniques that are statistically sound, accurate and repeatable. Those efforts should be the number one priority for every park and should be funded accordingly. As good stewards of national parks, we should do no less.

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Assembly of a Standardized Database for Vascular Plants in California's National Park Service Lands

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Abstract. We assembled a preliminary master inventory of vascular plants for California's 18 National Park Service units, using published checklists, inventory data provided by park staff, and the NPFLORA database. Because there is no uniform system for data collection or organization within the Park Service, we were unable to directly produce either summary or comparative analyses from the original data. The primary difficulty that we encountered was the lack of a uniform system of taxonomic nomenclature. We describe our methods for standardizing the data, and offer some preliminary summary statistics.

INTRODUCTION

Vascular plant inventories have been prepared for many conservation lands in the United States. However, with few exceptions such as the Natural Heritage Program (Jenkins, 1988; Shevock and Hennessey, 1986), these databases are organized as rather discrete entities. The data are stored in numerous formats, the taxa are ranked according to a variety of nomenclatural systems (Steffan, 1986), and as a result the data are relatively inaccessible (Loucks, 1986; Tangley, 1985). Such is the case with vascular plant inventories within the National Park Service's 18 California units. This lack of uniformity is not unexpected, considering that the data have been collected by a number of different botanists and assembled over a span of many years. The unfortunate result, however, is that regional summaries, comparative analyses, and other mixings and matchings of the data are quite difficult to prepare.

There is a growing need for summary information from biological inventory data, since many important conservation decisions will require a better understanding of the current biological status of protected lands (May, 1988; Shen, 1987; Wilson, 1985). Numbers and identities of species represent basic information useful in assessing the biological status of a park or, on broader scales, determining the extent to which parks and preserves sample the natural diversity that we rely on them to protect (Scott et al., 1987). This kind of information will be of most use when it has been compiled in a standard format, one that allows comparisons among different conservation lands or among protected and non-protected lands. On the basis of such comparisons, more detailed evaluations of conservation plans and objectives can be undertaken. In a pilot project to study methods for summarizing inventory data, we assembled lists of vascular plant taxa from 18 parks, monuments and recreation areas administered by the National Park Service in California. While these lands vary greatly in area and encompass a variety of habitats, they share a similar protected status. Because of that shared status, and because together they contain within their boundaries approximately 5% of the land mass of California (Anonymous, 1986), it is worth knowing how many and what kinds of species are found within their collective boundaries. We chose to work with species lists because they represent tractable, discrete, and relatively unambiguous biological information on which to build more detailed analyses (see Quinn and van Riper, this volume).

This paper represents a first scan of the available data. We stress that these data are preliminary, and present them for review and comment as well as for purposes of demonstration. While our ultimate goal is to draw general conclusions on the status and distribution of plant diversity in these National Park Service lands, we know that the data are incomplete relative to other public and private lands. Detailed analyses should await inclusion of forthcoming new data, as well as formal verification of the data presented here.

METHODS

Sources

Lists of vascular plant taxa were solicited from Park Service staff beginning in 1984. Fifteen units provided at least a basic checklist (Table 1), and published checklists were available in two cases (Anonymous, 1987; Norris, 1982). These original lists have been revised as more recent information was procured, and modifications will continue as more comprehensive inventories or monitoring programs are undertaken in various units. Thirteen of the parks are represented in the NPFLORA database (Waggoner and Thorwardson, 1983) which is being prepared in Washington, D.C. for the entire National Park System. We were provided with those data in electronic format, which greatly facilitated their incorporation into our own database. Four of the units (Yosemite National Park, Sequoia-Kings Canyon National Parks, Point Reyes National Seashore, and Pinnacles National Monument) have been the subjects of recent field inventories, and we will incorporate this new information as it becomes available.

Database Structure

Data are stored in dBASE 3+ files (Copyright 1987, Ashton-Tate, Inc.). The master file for vascular plants in the 18 parks currently occupies approximately 1200 Kb. This master list is structured to be simple and compact. It contains only family, genus, species and variety categories, the general origin of each taxon (native, introduced, cultivar or uncertain), an indication of rare or endangered status, and the Park Service units in which the taxon is reported to occur. The basic descriptor for each taxon is the combination genus-species-variety/subspecies classification, and the data may be ordered according to this descriptor or indexed by plant family name, as in a traditional checklist format. Full names are used in all cases. While codes and abbreviations are useful in reducing file sizes in plant databases (e.g., Lum, 1975), we found it more expedient to use full taxonomic epithets, particularly with respect to adjusting for differences in the nomenclatural systems used among the parks.

The subspecies and variety categories were often applied ambiguously, and in general may be formally indistinguishable (Lawrence, 1951). Therefore, we simply recorded subspecific and varietal epithets without differentiation. A related problem involved the use of intraspecific classifications that matched the species names. In some inventories, the species epithet was repeated for subspecies or variety, even though no subspecies or varieties are reported in the state of California. In other cases, when a particular species was known to incorporate a number of subspecific taxa, it was unclear whether an entry without a subspecific category indicated that identification stopped at that level, or that no particular subspecies or variety was known to occur in that park. For many analyses it is probably prudent to abandon taxonomic categories below the species level. However, this would constitute a significant loss of information on the California flora, in which about 30% of the plant taxa fall into subspecific categories (Smith and Noldeke, 1960; Howell, 1972). Consequently, our approach to this latter problem has been to use subspecific names only in cases when they do not match species names. This problem will be alleviated in the future by including the authors' names for all taxa, a practice now followed by several of the parks.

Classifications of origin (native, introduced, and cultivar) were assigned according to reference literature, and will in some cases require reclassification (M. Rejmanek, pers. comm.). Additional categories indicate special conservation status assigned to each of the taxa (Smith and Berg, 1988). In the future, the database will also record locations of voucher specimens maintained for each Park Service unit in which a taxon is known to occur (recommended in comments by D. Goforth, G. Fellers).

Common names may be quite useful in local and regional floras, but they were not useful to us. They are often duplicated among different scientific taxa, rarely apply below the species category, and add no information to the summary database. We therefore did not include them in our master list, although they are maintained in several individual park lists. In addition, a number of non-vascular plants were included in some of the individual park inventories, but we have restricted our presentation to vascular plants.

Standardization

The 15 checklists used to compile our database were organized in a variety of formats and, more importantly, used clearly different sources for assigning taxonomic classifications. In addition, none of them was fully compatible with its counterpart among the 13 NPFLORA database files (Table 1). Standardizing the entire data set to a single nomenclatural system was not a trivial process. Since this paper is presented in part to describe those efforts, we will explain them in some detail.

We first combined data for each park, removing obvious duplications, and these in turn were combined into a single master file, which at its initial stage held over 6200 entries. Within this master file, we determined which entries were taxonomic synonymies of others. This required a common reference. Unfortunately, until the completion of a comprehensive revision of Jepson (1925), a modern reference is unavailable, and so our nomenclatural designations were chosen to conform with Munz and Keck (1959) and Munz (1968). Identifying all of the synonymies required the use of local and regional floras (Gleason and Cronquist, 1963; Hitchcock and Cronquist, 1978; Jepson, 1925; Munz, 1974; Pusateri, 1963), as well as general reference works (Bailey et al., 1976; Kartesz and Kartesz, 1980; Mabberly, 1987). Taxa that had no apparent counterpart in Munz and Keck (1959) and Munz (1968) were classified according to Kartesz and Kartesz (1980). Presumed non-naturalized cultivars were classified according to Bailey et al. (1976). While there are obvious limitations to our overall approach, the structure of the resulting database is such that it can be easily modified to conform to updated classification systems.

The assignment of plant family classifications was more arbitrary. Names of many of the larger families are assigned as in Cronquist (1968), but in most cases, we simply deferred to Munz and Keck (1959) and Munz (1968). While this undoubtedly puts us out of step with current nomenclature, particularly among lower vascular plants, systematic changes in familial status can be easily made if they are needed to make the data more accessible.

As synonyms were identified, they were entered in a separate file, which includes a record number for each taxon that matches its "correct" counterpart in the master file. At this writing, our synonym file contains 684 entries which are cross-referenced to 643 taxa from the master file.

Individual lists for each of the 18 Park Service units were then compared with the synonym file and the master file. Scientific names were replaced as necessary to conform with the master database file. Finally, a park-by-park checklist was incorporated into the master file. These latter two steps were done using computer programs written in simplified relational database language.

RESULTS

General Summary

At this stage of data collection, we find 4558 taxa (3853 species) reported for the 18 California National Park Service units. Of these, 3958 (3276 species) are native and 473 (456 species) are introduced. We list an additional 119 taxa as cultivars, presumably planted by previous inhabitants or by park staff, and not properly naturalized. Eight of the taxa are of as-yet undetermined origin. A park-by-park checklist is available upon request from the authors.

We stress again the preliminary nature of this information. Many of the units may need more thorough field inventory work. For example, Channel Islands National Park contained 382 taxa according to the NPFLORA database, and 418 according to the list provided by the park in 1984. Combined, we found 454 reported taxa. Recent work (Anonymous, 1987) has nearly doubled the earlier inventories (Table 1). Basic field inventories continue in Death Valley National Monument and the Sequoia-King's Canyon complex, where some regions are to be more thoroughly studied (C. Douglas, pers. comm.; D. Graber, pers. comm.) At Pinnacles National Monument, where recent field collections are being verified, the number of taxa will be expanded by at least 35% (S. DeBenedetti, pers. comm.) Recent collections at Yosemite National Park, whose results are still in preparation, include approximately 75% more entries than the 751 which we are reporting (S. J. Botti, pers. comm.).

Finally, previously collected information from various parks may be in need of review, particularly since in many cases voucher specimens have not been collected to verify reported sightings (D. Graber, pers. comm.). This is certainly apparent from the fact that several of the taxa reported are presumed extinct in California (Smith and Berg, 1988).

TABLE 1. Preliminary numbers of vascular plant taxa for California's National Park Service lands.

Park	Area (km^2)	Park's listed taxa	NPFLORA Database taxa	Nomenclature Combined	Duplicates Removed	Total Species
CABR	0.6	95	195	219	200	186
CHIS	506	780[1]	382	857	780	751
DEVA	8368	1089[2]	1051	1231	1077	976
DEPP	3.3	232			232	229
FOPO	0.12	25			25	25
GOGA	177	489			489	474
JOMU	0.04	116[3]			116	115
JOTR	2270		684		684	631
LABE	187		280		280	277
LAVO	432	703	708	1041	769	730
MUWO	2.2	71			71	70
PINN	65	236	410	634	416[4]	396
PORE	275	725	732	784	746[5]	709
REDW	426	661	630	727	649	619
SAMO	607	940	811	1109	966	913
SEKI	3430		1194	1194	1184	1104
WHIS	172	18	472	472	472	448
YOSE	3080	44	751	751	751[6]	711
TOTAL	20194				4558	3851

* Park key: CABR = Cabrillo National Monument; CHIS = Channel Islands National Park; DEVA = Death Valley National Monument; FOPO = Fort Point National Historic Monument; GOGA = Golden Gate National Recreation Area; JOMU = John Muir National Historic Site; JOTR = Joshua Tree National Monument; LABE = Lava Beds National Monument; LAVO = Lassen Volcanic National Park; PINN = Pinnacles National Monument; PORE = Point Reyes National Seashore; REDW = Redwood National Park; SAMO = Santa Monica Mountains National Recreation Area; SEKI = Sequoia and Kings Canyon National Parks; WHIS = Whiskeytown National Recreation Area; YOSE = Yosemite National Park.

[1] Source: Anonymous (1987)

[2] Source: Norris (1982)

[3] Approximately 50 garden species listed by common name only, were not included.

[4] Upon completion of work in progress, this number should expand by about 35%.

[5] Recent inventory data is not yet included.

[6] Upon completion of work in progress, this number should expand by about 75%.

It is nevertheless noteworthy that the Park Service lands contain a large proportion of the California flora (approximately 65% of native vascular plant species, based on Howell [1972]), and undoubtedly more species will be listed upon completion of recent inventories. On the other hand, considering their geographic spread and the range of habitats covered, it is not surprising that as a group the 18 units are so taxonomically diverse. The proportion of introduced species (14%) is consistent with the overall estimates for California reported by Rejmanek et al. (1988), and this probably reflects the impacts of many past and present users of the parks (e.g., Vankat and Major, 1978; MacDonald et al., 1988). In comparison to the size of the database, the number of plant taxa of greatest concern for conservation seems disproportionately small. Our list includes 157 out of a potential 852 rare and endangered plants reported by Smith and Berg (1988). This may result from the fact that many rare species are restricted to specialized habitats not sampled by the eighteen units, or it may indicate a need for more field investigation within the parks. In either case, there is little doubt that the parks and monuments do serve an important role in harboring rare and endangered plants (e.g., Clark and Fellers, 1987).

DISCUSSION

There is growing recognition of the important role that National Park Service lands will play in the preservation of biological diversity, and that recognition is reflected in renewed efforts to improve the inventory and monitoring procedures in many of California's National Park Service units (various authors, this volume). We feel that these efforts would be conducted more efficiently and would provide information of greater immediate value if a standard format were adopted for all collection efforts (van Riper and Quinn, this volume; Cook et al., this volume; Sauvajot et al., this volume). Any effective state-wide database must conform to a single system of taxonomic nomenclature, cross-referenced to other sources. In addition to the Park Service lands, we are assembling available checklists for vascular plants of the California State Park system, as well as other conservation lands.

What we have learned at this point is that most of these individual inventories and checklists suffer from the same general lack of standardization. It is obviously not our purpose to criticize anyone for the current state of the data. The fact that so much recorded data exist is a tribute to the hard work of many dedicated individuals. Unfortunately, far too often their hard work is inaccessible, in the sense that each database stands alone.

We offer a preliminary standardized master list of vascular plants (available upon request from the authors), subject to review by all interested parties, as a first step in promoting a means to standardizing the collection and storage of regional plant inventory data for California parks and preserves. We hope that by publishing the data as they are, corrections and amendments can be made systematically, with the retention of a standardized nomenclature. Verification in many cases will require field work, and it is our additional hope that this master list will serve as a useful tool in the collection and preparation of future inventory data. Park biologists and other researchers will at least have available a master list that they can use to check off new entries in the field for a given park or location within a park. In park offices, entering basic information, such as the presence of previously unrecorded species, can be done with a few keystrokes; the master list itself can be expanded or linked to more detailed databases that incorporate habitat and range data. The master database itself would be maintained in a central location, and regular updates (including additions of new taxa) would be sent to each park, where the data could be restructured and modified to suit individual needs. In addition to improving the efficiency of field data collection, the use of a single shared database would allow regular analyses of the system-wide status of plant species diversity. The uses and benefits of standardized checklists are discussed at length by Quinn and van Riper (this volume).

Our principal contribution to this point has been the preparation of standardized data. Our choice of botanical nomenclature may be problematic for field biologists who prefer to use local floras that are more often up-to-date than Munz and Keck (1959) and Munz (1968). However, because we can provide a matching synonymy list that includes at least the majority of ambiguous taxa, this should not be a serious problem. That system of automatic cross-referencing may, in fact, be an important service of this project. The database will certainly be of greater value once it is rewritten to conform with new editions of Jepson (1925) and Kartesz and Kartesz (1980), both of which are scheduled for release early in the next decade.

A more fundamental task is the development of thorough, verified biological inventories for all of the parks. Fortunately, a great deal of work has already been done, and more is under way in California. Although the organization of comprehensive diversity data at a national scale will be delayed for some time, due to the fragmented nature and uneven quality of the component databases (Loucks, 1986; Shen, 1987), the National Park Service lands in California could serve as a model for those efforts. It is our hope that we can assist Park Service biologists in developing such a regional model for summarizing biodiversity data.

CONCLUSIONS

Biological inventory data is of greatest value when it provides comparative information, such as how many and what kinds of species are found in which locations, or how many species are found within a given set of parks or preserves. These kinds of questions can be addressed only when the data from the areas being compared conform to a common standard. Achieving uniformity in vascular plant inventory data for the National Park Service lands in California requires the adoption of a single arbitrary system of taxonomic nomenclature and the development of a companion list of nearly 700 synonyms.

Since we anticipate more and better data in the months and years to come, and since the California flora as a whole is being revised, we present very preliminary estimates of the numbers of taxa per park and the total represented in the entire 18-park system. More formal analyses, such as species-area relations, are inappropriate at this time. What is appropriate is a discussion of methods for the collection of inventory data and its incorporation into larger databases (see Quinn and van Riper, this volume). The methods that we have developed are designed to assist in the critically important work of gathering and summarizing biological inventory data, and we encourage advice and comments on our efforts. Copies of our master and synonym lists, subject to continuous updates and other modifications, are available on request.

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Toward An Inventory of Lichens of Pinnacles National Monument

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Abstract. Ninety-five species of lichens are recorded from Pinnacles National Monument and dominant lichens of the four principal plant communities are listed. Eight of these are fruticose species, 43 are foliose and 46 are crustose. One very rare species, *Texosporium sancti-jacobi*, is present. The lichens are abundant and in very good condition. Several species particularly sensitive to air pollution are well-developed and fertile. The lichen data, therefore, indicates that the air quality in Pinnacles National Monument is excellent. The most immediate threats to the lichens are fire and rock climbers.

INTRODUCTION

The biological resources of many plant groups, including the lichens, have not been documented in Pinnacles National Monument (PINN), San Benito County, California. It is somewhat surprising that there are so few collections of cryptogamic plants in a national monument as long-established as PINN. From a manager's perspective, such a lack of information is undesirable because one cannot adequately manage resources which are not known. Furthermore, the degradation of the environment or the success of current management programs cannot be assessed with any degree of accuracy. This research has focused on establishing a checklist of the lichens in PINN, concentrating on corticolous species because of their generally greater sensitivity to air pollutants. The dominant lichen species in the four major vascular plant communities were also evaluated.

METHODS

Historical Collections

The following herbaria were visited: San Francisco State University, University of California at Berkeley, Los Angeles Museum, and the

Smithsonian Institution. Enquiries were also made at the University of Colorado. The only herbaria that had collections from PINN were San Francisco State and the Smithsonian Institution.

Inventory

A general survey of the dominant lichens in the monument was conducted by sampling representative trees, shrubs, rocks and soils in the principal communities, viz., digger pine woodlands, deciduous and evergreen oak woodlands, cherry woodland, riparian and creek habitats, chaparral and rock outcrops. The following areas were surveyed (Fig. 1):

East side:

1. Bear Gulch Visitors Center. Bear Creek woodlands south to the Bear Gulch Caves and approximately one half mile down the creek.
2. Bear Gulch Reservoir. Along the south side of the reservoir and around Moses Spring.
3. Chalone Peak Trail to North Chalone Peak. From Bear Gulch Reservoir south to the summit.
4. Chalone Creek (north of the East Entrance Station road). Along the entire length of the main stream, including about one quarter mile up the Balconies Trail and the oak woodland flats west of Willow Spring.
5. Chalone Creek (south of the entrance road). Throughout the flats south of the road as far as where the ridges on opposite sides of the creek create a small gulch.

West side:

1. Chaparral campground area. Around the campground and the hills north and south of the Visitor Center.
2. The Balconies Trail. Down the Balconies Trail to the high point before it descends steeply toward North Chalone Creek.
3. The Fingers and the area north to the Chaparral Campground.
4. The Juniper Canyon Trail from Chaparral Campground to Scout Peak.
5. The large open meadow just inside the monument boundary. The grassland and the adjacent oak woodlands.

Frequency and Cover

The cover of all lichen species from ground level to eye height was recorded on the blue oak (*Quercus douglassii*) in three areas: Willow Spring, the oak woodland on the left after entering the west side of the

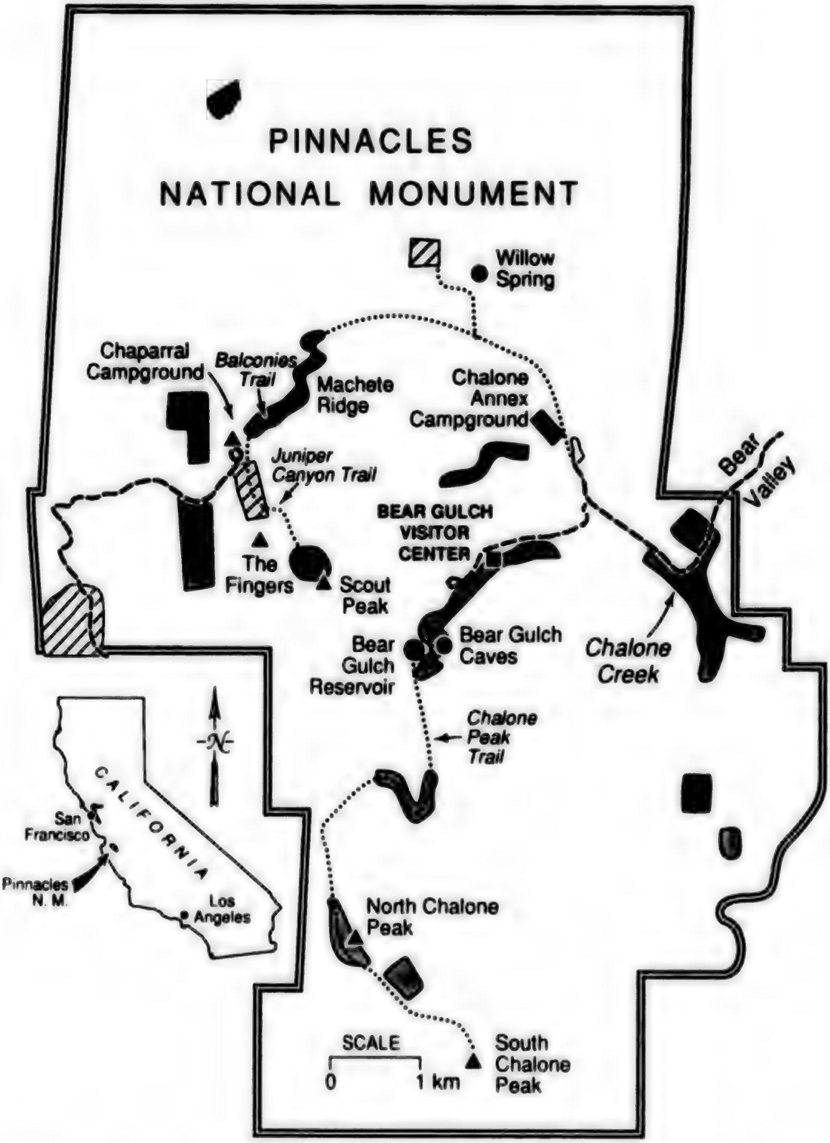

FIGURE 1. Study site locations in Pinnacles National Monument, California. Cross-hatched areas = quantitative study sites. Shaded areas = general survey sites.

monument, and the woodland just south of the Chaparral maintenance area. At least 25 trees were sampled in each area. No attempt was made to randomize the trees sampled because in most areas there were insufficient trees to allow for appropriate randomization. Notes were taken on the cover class using the Braun-Blanquet scale (Mueller-Dombois and Ellenberg, 1974), and on the fertility and condition of each lichen species. Frequency measurements were made using a band transect around each tree at breast height 1.5m above the midslope base of the tree. This latter technique is quite destructive; the soft, flaky bark is easily broken and the dry, fragile lichen thalli crumble when pressed.

RESULTS

History of Lichen Collecting in Pinnacles National Monument

No collections of lichens in PINN prior to 1963 were located. In that year, Dr. H.D. Thiers, San Francisco State University, visited the monument and made a number of collections then and on subsequent visits. In later years, several of his students also visited the area: J. Ammirati in 1966; D.E. Desjardin 1982, 1983; S. Hammer 1986; A.S. Methven 1982; S. Strick 1975; and B. Thiers 1977. Dr. Mason Hale, Smithsonian Institution, visited the monument on several occasions, making collections in 1965, 1978 and 1985.

Lichen Flora

The park hosts an abundant lichen flora, both in the number of species present and their biomass, which indicates a favorable environment. Ninety-five species of lichens are listed (Appendix I). With a few exceptions, the lichen flora, particularly the bark-inhabiting flora, consists of relatively widespread, common species. Eight of these are fruticose species, 43 are foliose and 46 are crustose. None of the specimens showed any signs of bleaching or other symptoms of air pollution damage.

Dominant Lichens by Habitat

The four principal lichen habitats in the monument are: xeric; riparian; chaparral; and foothill woodlands. All areas, except those recently disturbed by fire, flood waters, or human activity, support vigorous lichen communities.

Xeric habitat. Rock faces in most areas of the monument provide an extremely dry environment for plant growth. Lichens are abundant in this area, but several areas have been badly disturbed by rock climbers. The yellowish-green and red hues on the rocks, seen even from a distance, are a reflection of the dominance of lichens on this substrate. The diversity of species is high. The dominant species are *Acarospora chlorophana*, *Caloplaca saxicola*, *Dimelaena oreina*, *Rhizocarpon bolanderi*, *Tephromela atra*, *Xanthoparmelia conspersa*, *Xanthoparmelia mexicana*, and *Umbilicaria phaea*.

Riparian habitat. The dense vegetation along streams and wet gullies results in heavy shading which favors moss growth. Lichen diversity is high but only the following species are abundant: *Leptochidium albociliatum*, *Leptogium californicum* and *Leptogium lichenoides*. *Peltigera canina* is common in certain areas of Bear Gulch.

Chaparral habitat. Fire, on which chaparral is dependent, destroys all lichens. However, the decadent chaparral brush hosts a lichen community of low diversity but high biomass. The smaller branches are generally covered with the orange *Xanthoria polycarpa*, the yellow *Candelaria concolor*, and the black *Cetraria californica* and *Leptogium corniculatum*. *Evernia prunastri*, the most abundant species in the monument, is found on the larger branches and particularly at the base of the bush together with *Flavoparmelia caperata*, *Hypogymnia imshaugii*, and *Parmelina quercina*.

Foothill woodland habitat. The evergreen live oaks, digger pines, sycamore, manzanita, cherry and chestnut are poor hosts for lichens. Only two tree species, both deciduous oaks, had abundant lichen cover. The larger species, the valley oak (*Quercus lobata*) has a tall main trunk and a hard bark. Though lichen cover generally exceeded 50% on the east-facing side, the lichen diversity was low. The dominant species was *Physconia detersa*. The smaller blue oak is a good host supporting lichen cover up to 50% on the north-, east-, and south-facing trunk surfaces, as well as a reasonable diversity of lichens including *Candelaria concolor*, *Flavopunctelia flaventior*, *Leptogium* sp., *Melanelia glabra*, *Parmelia subrudecta*, *Physcia adscendens*, *Physcia biziana*, *Physconia detersa* (the dominant species on the trunks and larger boughs), *Ramalina leptocarpha*, *Ramalina menziesii* (the very obvious species hanging from the branches in areas subjected to coastal fog), *Rinodina exigua*, and *Xanthoria fallax*.

Quantitative Studies

Candelaria concolor and *Physconia detersa* were present in all transects in all three areas (Table 1). *Physconia*, a foliose species, had a high cover value whereas *Candelaria*, crustose species, had very low cover. *Rinodina exigua* and *Xanthoria fallax* were present in most transects but had low cover. *Flavopunctelia flaventior* was absent at the Chaparral site but had a high frequency with low cover at the other two sites. *Ramalina menziesii* is a species that hangs from branches. The bark of the blue oak is sufficiently rough, however, so that *Ramalina menziesii* fragments blown against the trunk are generally trapped there. The erratic frequencies in the three sites reflect the abundance of this species in the canopy.

However, the cover is always low because the larger thalli become too heavy and break away. All the other species display erratic frequencies averaging below 50% and have minimal cover.

TABLE 1. Frequency, cover, and cover class of corticolous lichens on blue oak at three sites in Pinnacles National Monument, California.
(Freq. = Frequency, Cov. = Cover class)

	Willow Spring		Chaparral		Soledad	
	Freq	**Cov**	**Freq**	**Cov**	**Freq**	**Cov**
Caloplaca ? holocarpa	.30	+	0.00	–	.30	+
Caloplaca sp.	.30	+	.80	+	.20	+
Candelaria concolor	1.00	+	1.00	+	1.00	+
Collema sp.	0.00	–	.10	+	0.00	–
Flavopunctelia flaventior	.80	+	0.00	–	1.00	+
Lecanora green	.40	+	.80	+	.20	+
Lecanora black	.20	+	.40	+	.10	+
Lecidia sp.	0.00	–	.60	+	.20	+
Leptogium furfuraceum	0.00	–	.10	+	0.00	–
Melanelia glabra	.40	+	.30	+	.70	+
Physcia adscendens	.20	+	.30	+	.30	+
Physconia detersa	1.00	1	1.00	1	1.00	1
Punctelia subrudecta	0.00	–	0.00	–	.40	+
Ramalina leptocarpha	.40	+	.60	+	.40	+
Ramalina menziesii	.10	+	1.00	+	.30	+
Rinodina exigua	.60	+	.90	+	.70	+
Xanthoria fallax	.80	+	.80	+	1.00	+

DISCUSSION

The Lichen Flora

It is unfortunate that crypotgamic botanists have ignored PINN in the past. There is an extremely poor historical record of the lichens. On the other hand, the isolation of the monument from human activity has resulted in the preservation of a good assemblage of lichens in abundance and in good condition. Approximately half of the lichens recorded (44) are also found in Sequoia and Kings Canyon National Parks (Wetmore 1985), illustrating the affinity of the lichen flora with the California foothills. Yet, some rare lichens are also present in PINN, e.g., *Texosporium sancti-jacobi*. This species is poorly known, perhaps because of its inconspicuousness and its extremely unusual habitat of old rabbit dung; a habitat that most lichenologists ignore.

The abundance and fertility of *Ramalina menziesii* was somewhat surprising. This West Coast endemic is normally confined to the coastal fog zone. It flourishes in PINN because the weather in the area mimics that of the coastal region approximately 35 miles to the west. The nights are generally cool, if not cold (Table 2); although coastal fog rarely reaches PINN, the meteorological conditions create a somewhat similar environment. The temperature falls below dew point during most evenings of the year. Most fruticose lichens are soft and damp every morning until about ten o'clock. The overnight dew and cool early morning weather create ideal conditions for lichen growth. However, none of the other coastal Californian endemics, e.g., *Niebla* spp, are found at PINN.

TABLE 2. Average monthly maximum and minimum temperatures (° F) in 1983 at Bear Gulch Weather Station, 398 m elevation, Pinnacles National Monument, California.

	Month											
	Ja	Fe	Mr	Ap	My	Jn	Jl	Au	Se	Oc	No	De
Maximum temperature	62	61	60	62	77	84	88	92	90	80	65	59
Minimum temperature	33	35	37	36	41	44	46	50	50	41	37	35
No. nights temperature does not reach dewpoint	3	0	0	3	0	11	15	16	19	9	3	1
No. days humidity did not exceed 75%	0	0	0	0	0	5	4	6	11	3	1	0

Air Quality Evaluation

Only one species, *Ramalina farinacea*, found in the monument is known to be very sensitive to air pollution, i.e., levels of sulphur dioxide with an annual average level below 50 micrograms per cubic meter (Wetmore, 1983). Four species are sensitive to concentrations around 50 $\mu g/m^3$, 13 between 50-100, two around 100, and two above 100 $\mu g/m^3$. The latter, *Buellia punctata* and *Lecanora muralis*, are considered tolerant.

The health of the lichens in PINN, as evidenced by their abundance, fertility, robust nature and lack of bleaching, indicates that the air quality is excellent. There is no evidence that sulphur dioxide pollution levels have reached even the lowest levels toxic (20 micrograms per cubic meter) to the most sensitive lichens. However, physical measurements of ozone on the east side of the monument indicate that levels have exceeded national standards (0.08 ppm) on at least five occasions in mid-afternoon between June and September 1987. The encroachment of smog from San Jose into the area around Hollister is cause for concern. The evidence from the Los Angeles Basin (Sigal and Nash, 1983; Neel, n.d.) indicates that smog would have a very severe impact on several of the lichens in PINN, e.g., *Ramalina menziesii*.

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APPENDIX 1

List of Lichens in Pinnacles National Monument

	*	*Acarospora chlorophana* (Wahlenb. ex Ach.) Massal.
		Aspicilia cinerea (L.) Körber
		Aspicilia gibbosa (Ach.) Körber
		Buellia disciformis (Fr.) Mudd
		Buellia lepidastra (Tuck.) Tuck.
T		*Buellia punctata* (Hoffm.) Massal.
		Caloplaca ferruginea (Huds.) Th. Fr.
		Caloplaca holocarpa (Hoffm.) Wade
		Caloplaca saxicola (Hoffm.) Nordin
		Caloplaca variabilis (Pers.) Müll. Arg.
S-I	*	*Candelaria concolor* (Dickson) B. Stein
		Candelariella spraguei (Tuck.) Zahlbr.
I-T		*Candelariella vitellina* (Hoffm.) Müll. Arg.
		Candelariella zahlbruckneri Hasse (Thomson)
	*	*Cetraria californica* Tuck.
	*	*Cladonia chlorophaea* (Flörke ex Sommerf.) Sprengel
		Collema sp.
		Cyphelium tigillare (Ach.) Ach.
	*	*Dermatocarpon miniatum* (L.) Mann
	*	*Dimelaena oreina* (Ach.) Norman
		Diploicia canescens (Dickson) Massal.
		Diploschistes scruposus (Schreber) Norman

APPENDIX I (continued)

I		*Diplotomma alboatra* (Hoffm.) Flotow
I	*	*Evernia prunastri* (L.) Ach.
I	*	*Flavoparmelia caperata* (L.) Hale
	*	*Flavopunctelia flaventior* (Stirton) Hale
	*	*Flavopunctelia soredica* (Nyl.) Hale
	*	*Heterodermia leucomelos* (L.) Poelt
	*	*Hypogymnia imshaugii* Krog
		Lecania fuscella (Schaerer.) Körber
		Lecania hassei (Zahlbr.) Noble
		Lecanora melaena (Hedl.) Fink
T	*	*Lecanora muralis* (Schreber) Rabenh.
I		*Lecanora pallida* (Schreber) Rabenh.
		Lecanora ? piniperda Körber
		Lecidea fuscoatra (L.) Ach.
		Lepraria caesioabla (B. de Lesd.) Laundon
		Leptochidium albociliatum (Desmaz.) M. Choisy
	*	*Leptogium californicum* Tuck.
	*	*Leptogium corniculatum* (Hoffm.) Minks
		Leptogium furfuraceum (Harm.) Sierk
	*	*Leptogium lichenoides* (L.) Zahlbr.
	*	*Letharia vulpina* (L.) Hue
	*	*Melanelia fuliginosa* (Fr. ex Duby) Essl.
	*	*Melanelia glabra* (Schaerer) Essl.
	*	*Melanelia incolorata* (Parr.) Essl.
	*	*Neofuscelia verruculifera* (Nyl.) Essl.
	*	*Nephroma helveticum* Ach.
		Ochrolechia upsaliensis (L.) Massal.
		Pachyospora verrucosa (Ach.) Massal.
I-T	*	*Parmelia sulcata* Taylor
	*	*Parmelina quercina* (Willd.) Hale
	*	*Peltigera canina* (L.) Willd.
	*	*Peltula euploca* (Ach.) Ozenda and Clauz.
		Pertusaria chiodectonoides Bagl. ex Massal.
	*	*Physcia adscendens* (Fr.) H. Olivier
I	*	*Physcia aipolia* (Ehrh. ex Humb.) FÜrnr.
	*	*Physcia biziana* (Massal.) Zahlbr.
	*	*Physcia callosa* Nyl.
	*	*Physcia phaea* (Tuck.) Thomson
I	*	*Physcia stellaris* (L.) Nyl.
I	*	*Physconia detersa* (Nyl.) Poelt
I	*	*Physconia distorta* (With.) Laundon
I	*	*Physconia grisea* (Lam.) Poelt
		Porpidia zeoroides (Anzi) Knoph and Hertel
		Psora sp.
I	*	*Punctelia subrudecta* (Nyl.) Krog
S	*	*Ramalina farinacea* (L.) Ach.
		Ramalina leptocarpha Tuck.
	*	*Ramalina menziesii* Taylor
		Rhizocarpon bolanderi (Tuck.) Herre
		Rhizocarpon geographicum (L.) DC.

APPENDIX I (continued)

I		*Rinodina exigua* (Ach.) Gray
		Rinodina tephraspis (Tuck.) Herre
		Sarcogyne clavus (DC. in Lam. and DC.) Krempelh.
	*	*Sticta fuliginosa* (Hoffm.) Ach.
		Tephromela atra (Huds.) Hafellner
		Texosporium sancti-jacobi (Tuck.) Nádv.
		Thelomma mammosum (Hepp. in Hartung) Massal.
		Thelomma occidentale (Herre) Tibell.
		Toninia caeruleonigricans (Lightf.) Th. Fr.
		Toninia massata (Tuck.) Herre
		Trapeliopsis granulosa (Hoffm.) Lumbsch
		Tuckermannopsis canadensis (Räsänen) Hale
	*	*Tuckermannopsis merrillii* (Du Rietz) Hale
	*	*Umbilicaria phaea* Tuck.
S-I	*	*Usnea subfloridana* Stirton
		Verrucaria aethiobola Wahlenb. in Ach.
	*	*Xanthoparmelia conspersa* (Ehrh. ex Ach.) Hale
	*	*Xanthoparmelia cumberlandia* (Gyelnik) Hale
	*	*Xanthoparmelia mexicana* (Gyeln.) Hale
	*	*Xanthoparmelia taractica* (Kremplh.) Hale
I	*	*Xanthoria candelaria* (L.) Fr.
S-I	*	*Xanthoria fallax* (Hepp. in Arnold) Arn.
I	*	*Xanthoria polycarpa* (Hoffm.) Rieber

S = sensitive to air pollution levels of sulphur dioxide with an annual average level below 50 micrograms per cubic meter.
S-I = sensitive to concentrations around 50 μg/m^3.
I = intermediately sensitive to concentration between 50-100 μg/m^3.
I-T = sensitive to concentrations around 100 μg/m^3.
T = tolerant of concentration above 100 μg/m^3.
* = treated in Hale's (1979) "How to Know the Lichens."

Alien Plant Inventory of Golden Gate National Recreation Area

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Abstract. An extensive field survey in the Marin Headlands District of the Golden Gate National Recreation Area identified major invasive plant populations and described biological and environmental parameters of each population site. Information on 22 alien (i.e., exotic) species encountered were stored on a computer database and keyed to field maps for easy manipulation. A discussion with specific examples illustrates how the information has been used: (1) to set priorities to control alien plant populations; (2) to evaluate eradication techniques; (3) to monitor changes in alien plant populations; and (4) to provide a better understanding of the environmental factors that limit the extent of individual alien species.

INTRODUCTION

The alien plant problem at Golden Gate National Recreation Area (GGNRA) is substantial. The park is on the edge of three urban San Francisco area counties. Decades of grazing, agriculture, fire suppression, army occupation, trampling and alien plant invasion from the urban area have changed the pristine nature of the park (Howell, 1982). The species composition of native plant communities changed to include, and in some cases are dominated by alien species.

Despite the proximity of GGNRA to urban areas, National Park Service management policies require managers to conserve and protect the native flora and fauna and to remove alien species when they threaten the natural resources (USDI, 1988). An important first step in the effective and efficient management of alien plant infestations is to conduct a baseline inventory leading to a management strategy. Specific objectives of the baseline inventory are: (1) to gather biological and environmental information necessary for setting priorities for systematically controlling alien plant populations; (2) to evaluate the success of eradication techniques;

(3) to monitor changes in alien plant populations; and (4) to identify research needs and provide an understanding of the environmental factors that affect the distribution of individual alien species.

This paper describes the techniques used in gathering baseline data and describes the results in a general manner. The focus of the paper is to display the importance of the data to management and therefore not all of the extensive data collected is displayed. Examples are used to describe each of the ways such data can benefit park operations and facilitate decision-making. Four species are used to discuss the database relative to the objectives described above.

METHODS

Field Survey

A standardized alien plant field survey form was developed. The survey was conducted from early May to mid-July 1987, covering approximately 9500 acres of GGNRA. This area was divided into blocks which were each systematically surveyed. Floras were used to confirm the species noted were alien (Munz and Keck, 1968; Howell, 1970). All alien species except grasses were included in the survey. Aliens within dense patches of brush and poison oak (*Rhus diversiloba*) were surveyed with binoculars.

Each alien plant population was mapped, in the field, on a large-scale aerial photo and also on a 7.5-minute topographic map. A planimeter was used to estimate the population area. Numbers of alien plants in each population were estimated by counting individuals of each alien species in a representative sub-sample and then extrapolating to the entire population. Population spread was estimated by the presence of young plants adjacent to mature plants.

Environmental parameters at each site were noted in a general context. Soil depth was recorded as shallow, deep or unknown and soil moisture as wet, dry or unknown based on the Soil Survey of Marin County (Kashiwagi, 1985). Sunlight was noted as full, filtered or shade. Slope aspect and topographic position were recorded, as well as any associated disturbance (trail edges, grazed land, roadsides, erosional slump, landfill or any other disturbance observed).

A particular plant of concern is striata broom (*Cytisus striatus*). This broom is a new species to Marin County and is not listed in any of the California state floras. The species was seeded onto a road bank for landscaping in 1966, and appears to be spreading rapidly. The extent of the initial seeding is unknown. However, it is clear that populations away from the immediate road corridor are not part of the original seeding project. Each of these populations were identified and numbered. Striata broom populations were visited twice to estimate spread. Shrub ages were estimated for selected populations by counting annual rings. Distances from the original seeding were estimated from field maps. To obtain an estimate of the rate of spread of the species, the distance of each population from the original seeding was divided by the number of years that lapsed between the original seeding and the establishment of the plants in each of the additional populations.

RESULTS

Twenty-two species were identified as invasive aliens in this study (Table 1). Alien species that infected the greatest area include: Italian thistle (*Carduus pycnocephalus*), which is present on 628.7 acres; poison hemlock (*Conium maculatum*) present on 515.4 acres; and milk thistle (*Silybum Marianum*) present on 249.9 acres (Table 1). Field mustard (*Brassica campestris*) and French broom (*Cytisus monspessulanus*) are each present on more than 150 acres. Species that are present on more than 50 acres include: fennel (*Foeniculum vulgare*), eucalyptus (*Eucalyptus globulus*), Monterey pine (*Pinus radiata*), Monterey cypress (*Cupressus macrocarpa*), Scotch broom (*Cytisus striatus*), sea fig (*Carpobrotus chilense*), thoroughwort (*Eupatorium adenophorum*), and pampas grass (*Cortaderia jubata*).

TABLE 1. Alien species of concern in the Marin Headlands District of GGNRA.

Species	**Total Area Affected (acres)**
Poison hemlock (*Conium maculatum*)	515.4
Italian thistle (*Carduus pycnocephalus*)	628.7
Fennel (*Foeniculum vulgare*)	129.0
Field mustard (*Brassica compestris*)	207.6
Milk thistle (*Silybum Marianum*)	249.9
Gorse (*Ulex europaeus*)	0.006
Eucalyptus (*Eucalyptus globulus*)	83.4
Monterey cypress (*Cupressus macrocarpa*)	51.2
Monterey pine (*Pinus radiata*)	78.5
Black acacia (*Acacia melanoxylon*)	16.7
French broom (*Cytisus monspessulanus*)	161.5
Striata broom (*Cytisus striatus*)	44.4
Scotch broom (*Cytisus scoparius*)	94.6
Sea fig (*Carpobrotus chilense*)	60.8
Thoroughwort (*Eupatorium adenophorum*)	52.4
Pampas grass (*Cortaderia jubata*)	58.7
Cotoneaster (*Cotoneaster sp.*)	45.8
African capeweed (*Arctotheca calendula*)	14.5
Calla lily (*Zantedschia aethiopica*)	5.4
Water hyacinth (*Eichhornia crassipes*)	0.25
German ivy (*Senecio mikanioides*)	15.9
English ivy (*Hedera Helix*)	0.05

The following results apply to the examples discussed in the Management Implications section.

Striata broom spread as much as 170 feet/year since the original seeding in 1966 (Table 2). Plants appear to establish more easily on disturbed or rocky sites. The largest population (#9) is contiguous to the original seeding and is spreading into grassland areas. The second largest population (#3) was on a disturbed graded site. The third largest population (#7) was on a rocky windblown ridge. The population invading the grassland (site #9) had a greater range of ages and were older, on the average, than other populations, indicating a longer period of establishment and opportunity for spread.

TABLE 2. *Cytisus striatus* sites including number of individuals, age and distance and direction of sites from original seeding (site 10).

No.	Distance (D) From Original Population (ft)	Number of Individuals	Average Age of Plants (yr)	Direction From Orig. Population	Spread Per Year (ft)
1	2100	1		SE	105
2	800	7	4.8	SE	40
3	2300	450	5.3	S	115
4	1900	1		S	95
5	1900	1		S	95
6	500	3		S	25
7	1000	18	5.1	S	50
8	3400	1		SE	170
9	100		9.2	W	5
10	0				
11	3000	1		SW	150
12	2900	1		W	145
13	600	1		NW	30
14	1200	4		N	60
15	1300	1		N	65
16	1000	1		N	50
17	1000	3		N	50

The population spread at an average rate of 94 feet/year to the south, 75 feet/year to the west and 42 feet/year to the north. No spread to the east was observed. The 483 plants established to the south, 10 established to the north, and one plant appeared to the far west.

Baseline data shows how each species is or is not restricted by several environmental factors. Sweet fennel (*Foeniculum vulgare*) is used here as an example.

Seventy-four populations of fennel were located in the Marin Headlands. Soil moisture and soil depth appeared to be associated with the distribution of this species. Seventy-two of the 74 populations (97%) were located on shallow soil and sites of low soil moisture. Only two populations (3%) were on sites with moist, deeper soils.

Disturbance was also related to the presence of fennel (Table 3). Eighty-one percent of the populations were on disturbed sites that were associated with construction of some sort. The major disturbance with which these plants were associated was road edges (62%), followed by trail edges (9%), batteries (5%), parking lots (3%) and an old home site (1%). No populations were associated with grazing or erosion.

Eradication techniques used to control pampas grass (*Cortadaria jubata*) are evaluated from the data collected (Table 4). Manual removal was 95% successful, whereas herbicide treatment using Roundup (glyphosate), applied in the fall, was 84% successful after three treatments.

TABLE 3. Fennel Populations related to disturbance in the Marin Headlands.

Total Number of Populations	**74**	
Number Associated With Disturbance	**60**	**(81%)**
Road edges	46	(62% of total number of populations)
Trail edges	7	(9% of total number of populations)
Batteries	4	(5% of total number of populations)
Parking lots	2	(2% of total number of populations)
Home site	1	(1% of total number of populations)
Total	60	(81%)

DISCUSSION AND MANAGEMENT IMPLICATIONS

The alien plant inventory database benefits park natural resources management operations and facilitates decision-making in at least four ways: setting priorities for alien plant control, documenting future alien population spread, understanding the environmental parameters that limit alien plants, and evaluating the success of eradication techniques.

Setting Priorities for Control

The number, cover and area data furnish the information needed to set priorities for removal and to estimate costs. Maps allow managers to see, at a glance, the extent of each problem species.

The 7.5-minute quadrangle map generated for French broom allows the manager to see the distribution and general extent of the 112 populations that exist in the Marin Headlands. An additional large aerial photograph

TABLE 4. History of some pampas grass populations in the Marin Headlands. All Roundup applications were at 2% solution.

Site no.	Date	Treatment	No. of Plants
97	10/86	pulaski	20
97	9/7	Roundup	20
444	1/85	cut off flowers	approx. 500
444	1/85	Roundup	approx. 500
444	10/86	respray Roundup	approx. 500
444	11/86	respray Roundup	several
444	6/87	none	
495	11/84	cut and sprayed	5
495	5/86	cut w/pulaski	5
495	6/87	none	1
496	1/85	cut and sprayed	800
496	10/86	Roundup	800
496	11/86	respray Roundup	?
496	6/87	none	200
497	10/86	cut and sprayed	600
497	6/87	none	95
542	12/86	chop out	60
542	6/87	none	2

with three overlays includes all alien species. This map allows managers to see how French broom populations are distributed in relation to other alien plant species populations. These visual displays are indispensable for identifying projects quickly when an unanticipated volunteer group is available for work.

The maps also allow managers to set up systematic control strategies for each alien species. Knowing the locations of large, established populations and small, maverick populations allows managers to determine what the most effective removal sequence would be. For example, managers may choose to first remove plants that affect wildland areas, and work backwards toward populations most associated with more disturbed environments.

Documenting Future Population Spread

Management staff uses these data to anticipate spread rates and direction, and to identify priorities for treatment. Major spread of striata broom was to the south and southeast (Table 2) probably due to the prevailing wind. This inventory data has facilitated the following analysis: since spread to the north of the original seeding is limited and the spread to the south is great, the 10 plants north of the original seeding should be removed to prevent the buildup of a seed source. This will avoid a potential problem of plant spread to the southwest, southeast and south of these plants.

Treatment of the southern population takes a different course. By working from the southernmost populations back toward the original seeding area, southward spread is minimized.

Intensive monitoring to detect maverick plants will be concentrated in areas south of the original seeding where the rate of spread is fastest. Northern areas will be monitored less frequently since spread to those areas occurs more slowly.

Understanding the Environmental Parameters of Each Species

Knowledge of the environmental factors involved in fennel distribution is valuable when construction projects are planned. When such projects are planned on shallow, dry soils, the occurrence of fennel will be anticipated and maintenance costs associated with immediate removal of young plants will prevent a larger problem in the future. Such disturbance on deeper or moister soils need not consider the invasion of fennel as a potential problem. Disturbance not associated with construction, such as grazing and natural erosion, need not be considered as having a potential major fennel problem.

Evaluating Success of Eradication Techniques

The alien plant database includes a detailed recording of each treatment on any population. The Marin Headlands District has worked to eliminate pampas grass for several years using manual and herbicide treatments (Table 4). A 2% solution of glyphosate (Roundup) was used in November 1984 and January 1985. The treatments resulted in partial eradication in the four areas.

A third treatment of glyphosate occurred in October 1986, using a large sprayer mounted on a truck. Plants still green in November were sprayed again. This double treatment was done on approximately 1900 plants and was 84% successful, with only 262 plants found in 1987.

Digging plants out with a pulaski was 95% successful on the 60 plants treated in December of 1986, with two plants remaining in June 1987. Although this treatment is effective, it is very labor-intensive.

Managers can evaluate treatment effectiveness by examining this type of data and making appropriate changes. In this case, manual treatment was successful but labor-intensive. The use of a backhoe, or small heavy equipment such as a bobcat, could be substituted in some cases. These could only be used in areas that are already disturbed.

Herbicide treatment, in the fall, with 2% glyphosate is effective. Herbicide treatment just after flowering, when the plants are more metabolically active, may prove to be more successful. Application techniques should be thoroughly reviewed and defined for effective treatment.

CONCLUSION

The examples noted here illustrate the variety of uses the inventory and monitoring data has to managers. The database enables managers to base decisions on observational data that can help set appropriate priorities and identify projects. It allows for less repetition of mistakes. This results in efficient management of manpower and resources.

The alien plant database is used to define management programs, to respond to current events by documenting the extent of the problem, to aid local alien plant research efforts, to interpret ecological problems and to build a volunteer alien plant removal team in the Marin Headlands. The volunteer effort includes treatment and monitoring of populations, and education of the public.

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Establishment and Control of Bull Thistle (Cirsium vulgare) in Yosemite Valley

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Abstract. *Cirsium vulgare* is an alien biennial that has become established in meadows in Yosemite Valley. It is classified as one of the park's three worst weeds. Thistle germination and seedling survival were studied in plots along a moisture gradient. The plots, originally free of thistles, were either left as controls, cleared of vegetation and litter, or cleared of vegetation and litter and the soil disturbed. Two hundred thistle seeds were then placed on each plot and the plots censused for thistles monthly. Germination was promoted by removal of vegetative cover and further promoted by soil disturbance. Establishment of thistles was greatest on plots with intermediate levels of soil moisture. Experiments in which adult thistles were cut at the soil surface were conducted to determine whether date of cutting or removal of the cut plants from the area influenced the number of thistles resprouting. Differences between the cutting treatments were not significant and date of cutting had no detectable influence on probability of resprouting. Over 80% of adult thistles originally present in control plots survived the summer but less than 2% resprouted in plots where cut plants were removed while slightly more than 5% resprouted in plots in which cut plants were not removed. Means of height and number of flower heads produced were lower for plants that resprouted (44 cm and 3.7 flowerheads) than for adults in the control plots (85 cm and 15.8 flowerheads).

INTRODUCTION

Statement of Problem

Cirsium vulgare (Savi) Ten., the bull thistle, is an alien weed that has become established at lower elevations in Yosemite National Park. It is common in meadows on the floor of Yosemite Valley and is classified as one of the park's three worst weeds (National Park Service, 1985). Alien plants capable of reproducing and spreading are undesirable in a national park because they compete with native species and alter the natural scene.

National Park Service policy dictates that populations of alien species be controlled and, where possible, eradicated (National Park Service, 1978). However, control of bull thistle in Yosemite National Park has proven to be especially difficult. The presence of native *Cirsium* species in the park makes it unlikely that a biocontrol agent could be found which would attack only bull thistle (Andres, 1980). Work parties pulled and/or dug thousands of thistles out of the meadows each summer in the 1960s and early 1970s but the populations rebounded after the practice was discontinued (National Park Service, 1985; L. West, personal communication).

The problem was compounded several years ago when thistle numbers in one meadow appeared to increase substantially following a prescribed burn. Since that time prescribed burns in the meadows have been strictly limited. The burns were conducted, in part, to kill young trees, particularly ponderosa pines (*Pinus ponderosa* Dougl. ex P. and C. Lawson), that would otherwise colonize the meadows. Ponderosa pines generally become too tall and/or well insulated by bark to be killed by a burn by the time they are five to 10 years old (J. van Wagtendonk, personal communication). Extensive meadows existed in Yosemite Valley when Euro-Americans first saw it in the mid-1800s and had probably been maintained for centuries, at least in part, by burning conducted by native Americans (Ernst, 1943; Gibbens and Heady, 1964; Heady and Zinke, 1978). If the meadows were taken over by pines, most meadow species would be unable to grow in their shade and the spectacular views of the surrounding waterfalls, granite cliffs and mountains many park visitors come to see would be obscured. Thus, the Park Service has a dilemma on its hands. However, despite the circumstantial evidence, it is not known if prescribed burns indeed promote bull thistle populations.

Experiments designed to determine the response of bull thistle populations to prescribed burns have been begun and the first test burn was conducted in March 1989. Here, results of studies on effects of two other types of disturbance (removal of vegetative cover and soil disturbance) and of soil moisture on thistle establishment are discussed. Results of a short study on cutting adult plants at the soil surface before they flower (rather than digging their roots out) in order to kill them and control their populations are also discussed.

Background

Cirsium vulgare is a member of the sunflower family (Compositae) native to Europe and Asia Minor. It was probably introduced to eastern North America during colonial times (Moore and Frankton, 1974) and first appeared in California in the San Francisco Bay area sometime in the early 1900s (Jepson, 1911). It is now found at elevations up to 2750 m (9000 feet) in California and is common in waste places, pastures and mountain meadows below 1520 m (5000 feet) in the northern and central portions of the state (Munz and Keck, 1959). It is generally classified as a biennial, but some individuals flower in their first year while others require four to five years (Klinkhamer et al., 1987; de Jong and Klinkhamer, 1988a). In montane California bull thistle sets seed in late summer and early fall. The seeds may germinate in autumn after the onset of rains or in the spring after the snow melts. The plants produce a rosette of spiny leaves in their first year and generally overwinter as rosettes. Typically, they bolt and produce flowering stalks 0.5 to 1 m high in the second year and flower sometime between late June and October (Munz and Keck, 1959). The number of inflorescences per plant varies widely, but individuals with more than 60 are common in Yosemite Valley. A single inflorescence may produce more than 250 seeds (van Leeuwen, 1983). Each seed is loosely attached to a feathery pappus and well suited for wind dispersal. The plants reproduce only by seed and die after they set seed.

The factors favoring germination and successful growth of *Cirsium vulgare* in Yosemite Valley are unknown. Competition studies indicated that the species does especially well under high nutrient conditions (Austin et al., 1985). Grime (1979) classified it as a species of disturbed habitats and work in pastures and adjacent areas in Australia indicated it is promoted by grazing which reduces competition for light and water by grasses and forbs (Forcella and Wood, 1986). George et al. (1970) found that seedling establishment was greater in disturbed areas than in areas with full vegetative cover. Klinkhamer and de Jong (1988) studied thistles in a coastal dune area in the Netherlands and found that seedling establishment was greater in plots from which vegetative cover was removed and the soil disturbed than in control plots. They found no differences in seedling mortality between disturbed and undisturbed plots. Van Leeuwen (1981) found seedling mortality higher in undisturbed sites, however, and hypothesized that seeds there were forced to germinate under unfavorable conditions by microbial activity, while seeds in disturbed sites were less influenced by microflora. Klinkhamer and de Jong (1988) pointed out that their disturbances were small (30 × 30 cm) and noted that the size of a disturbance may determine its effects on seedling establishment. Observations of poor seedling survival on open sandy sites led them to suggest that removal of vegetative cover over large areas may negatively effect establishment. Small openings in the vegetation were common in their study area due to trampling and the activities of rabbits, pheasants and moles. Small-scale disturbances caused by human trampling and gopher digging are common in Yosemite Valley's meadows.

Soil moisture is another factor likely to influence bull thistle establishment in Yosemite Valley. De Jong and Klinkhamer (1988b) reported a positive correlation between seedling survival and water content in the top 10 cm of the soil in the coastal dune area of the Netherlands mentioned above. They also found that year to year variations in seedling survival were correlated with summer rainfall and that survival and growth increased in artificially watered plots. Soil moisture levels below 10 cm were apparently high enough year round to allow plants to draw water in this area and soil water may not have been a critical factor once the root developed.

Bull thistle is a serious pest in pastures in Australia and North America and thus the target of many biological, chemical and mechanical control efforts. It is attacked by many insect species in Europe where it is native. One of these species, *Urophora stylata* F., a gall-forming fruit fly, was introduced as a biocontrol agent in Canada (Harris and Wilkinson, 1984). Biocontrol has not been attempted in California, however, because of bull thistle's status as a minor pest there and the potential for damage to native *Cirsium* species, some of which are rare and endangered (Andres, 1980). The need for caution was emphasized by observations of *Rhinocyllus conicus* (Froelich), a weevil introduced to control alien thistles, feeding on the natives *Cirsium californica* Gray and *C. proteanum* J. T. Howell in southern California (Goeden and Ricker, 1986). Several herbicides, including 2,4-D and MCPA, are effective means of controlling bull thistle populations (Lorenzi and Jeffery, 1987; Burrill et al., 1987). The National Park Service has banned 2,4-D, however, and allows use of herbicides to control alien species only as a last resort (National Park Service, 1985). Harris and Wilkinson (1984) stated that bull thistles can be controlled effectively by mowing shortly before they flower, but cautioned that they will resprout and flower if mown too early. McCarty and Hattling (1975) found that a related species, *Carduus nutans* L. (musk thistle), will not set seed or resprout if mown within two days of anthesis, while plants mown four days after anthesis produce substantial amounts of seed. The uneven flowering times of the plants made it necessary to mow repeatedly. The National Park Service cannot mow meadows where bull thistle is a problem but work crews could cut individual thistles at or near the soil surface. This would likely be quicker and cause less soil disturbance than pulling or digging them up and, if effective in killing them, would be a more efficient method of control.

STUDY AREA

The experiments described here were conducted in meadows in Yosemite Valley in the central Sierra Nevada of California (Fig. 1). The Valley is approximately 11 km long and 1 km wide and its floor is at an elevation of approximately 1220 m (4000 feet). The area has a Mediterranean-type climate with hot, dry summers and cool, wet winters.

Descriptions of the meadows may be found in Ernst (1949), Gibbens and Heady (1964) and Heady and Zinke (1978). They cover an estimated 150 hectares bordering the Merced River or its tributaries and are relatively flat with slopes of less than 3%. Soils are principally derived from alluvial sands and have not developed mature profiles, although some are rich

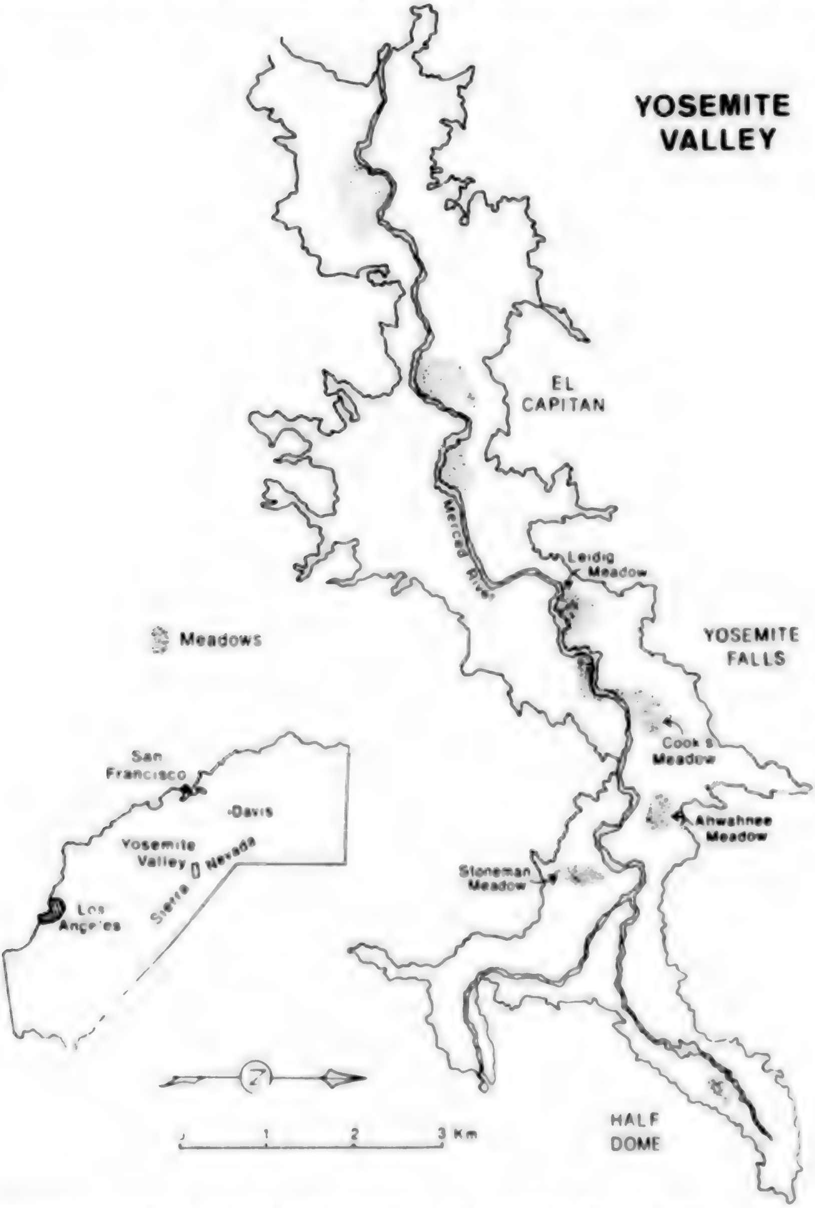

FIGURE 1. Map of Yosemite Valley indicating the location of the study sites.

in organic matter. Differences in vegetation within the meadows appear to be determined by drainage or depth to water table. The wettest areas in sloughs and old meander scars are dominated by *Carex vesicaria* L. and *C. senta* Boott. Intermediate areas are often dominated by other *Carex* species, *Poa pratensis* L. or *Agrostis alba* L. and drier sites by *Elymus glaucus* Buckl., *Lessingia leptoclada* Gray and *Bromus tectorum* L. A number of other species, including several grasses, *Pteridium aquilinum* (L.) Kuhn (bracken fern), *Heracleum lanatum* Michx. (cow parsnip), *Apocynum cannabinum* L. (dogbane) and bull thistle are conspicuous in some meadows. The meadows were significantly altered within the last 120 years by human activities, including ditching to improve drainage, plowing and use for crop production and pasturage. Today they are subject to locally heavy trampling by park visitors and horses (Foin et al., 1977).

METHODS

Disturbance Experiment

An experiment to determine the influence of disturbance and soil moisture on bull thistle germination and establishment was initiated in November 1987 in a meadow that was relatively free of the species. Three approximately parallel transects were run in Liedig meadow from low, relatively wet areas in an old meander scar to higher, drier areas with relatively sparse vegetation. Five sites were chosen along each transect and at each site, three 0.5 × 0.5 m (0.25 m^2) plots were marked out (45 plots total). At each site three treatments were randomly assigned to the three plots: (a) control; (b) vegetation clipped to the soil surface; and (c) vegetation clipped and the soil turned over to a depth of 15-20 cm. Two hundred bull thistle seeds were then scattered on each plot (9000 seeds total). The seeds had been gathered from plants in a nearby meadow two months earlier. Because this work was conducted in an area nearly free of adult thistles, it was assumed that few, if any, viable *Cirsium vulgare* seeds were present on the plots at the start of the study and that any present would be of negligible importance relative to those added. The bull thistles on each plot were censused at monthly intervals through August 1988. The plots were also checked every second week after the May census to determine if additional germination occurred. Differences in the number of plants that germinated and survived between treatments were compared with ANOVA.

Soil moisture was determined gravimetrically from soil cores taken at depths of five to 10 and 15 to 20 cm from each site in order to obtain a rough measure of relative availability of water. Soil cores were collected during the annual dry season in early July.

Manual Control Experiment

Experiments in which adult (bolted) thistles were cut at the soil surface were conducted in Ahwahnee meadow to determine whether date of cutting or removal of the cut plants from the area had an influence on the number of thistles resprouting. Three sites with heavy thistle infestations were chosen in late June 1988 and ten 2 × 2 m plots set up at each site (30 plots total). Four plots at each site were assigned a treatment in which all thistles were to be cut at the soil surface and removed from the area (cut and remove) and four other plots were assigned a treatment in which all thistles were to be cut at the soil surface and left on the plot (cut and leave). The two remaining plots at each site were controls. The treatments were administered to one plot at each site in late June, and to other plots in mid-July, late July and mid-August. All thistles in the plots were classified as bolted adults, rosettes or seedlings and the numbers of each type were recorded before they were cut with hand clippers or a machete. Thistles in the control plots were classified and counted in mid-July. Adults and rosettes on all plots were recounted in mid-September and the proportions of plants that had survived or resprouted since earlier counts were calculated. These data were arcsin-transformed and differences between treatments were then compared with ANOVA. Pairwise comparisons between treatments were made with Tukey's w procedure. The number of flowerheads on and height of each adult thistle were also recorded in September.

RESULTS

Disturbance Experiment

Removal of vegetation promoted successful establishment of bull thistle rosettes and soil disturbance further promoted it. The seeds did not germinate until mid-May and from that time on, more were present in the clipped and dug plots than in the clipped plots which in turn had more than the control plots (Fig. 2a). Less than 10% of the seeds (257 of 3000 sown) germinated and survived long enough to be counted even in the clipped and dug plots. A few germinated on the plots in early June and were identified as new plants in the June census, but no new germination was detected thereafter. When plots were blocked by site, ANOVA indicated that the differences in numbers of thistles between treatments were significant shortly after germination in May ($p < 0.01$) and through the summer months (June, $p < 0.05$; July and August, $p < 0.025$). Five thistles in clipped and dug plots bolted and began flowering by late August 1988, less than nine months after the seeds were planted. No bolted plants were found in clipped or control plots. It is important to note that a few individuals became established in control plots in which there was no detectable disturbance of the vegetation or soil.

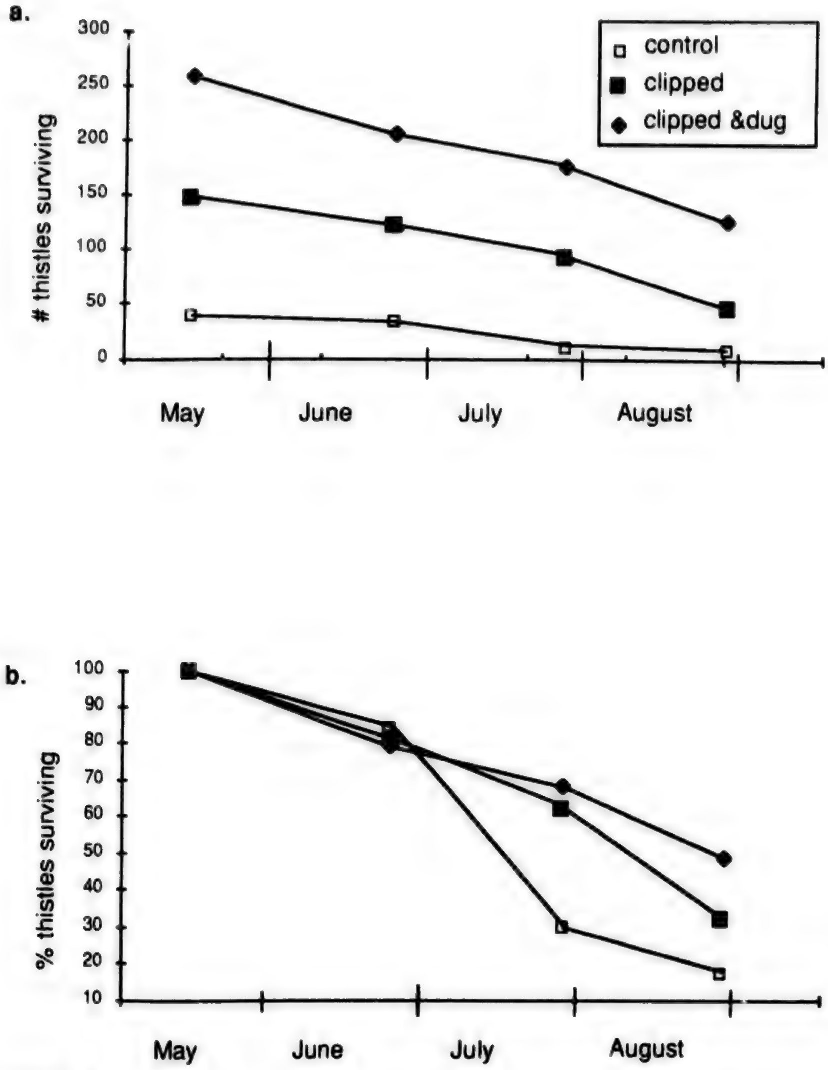

FIGURE 2. Disturbance experiment.

a. Total number of seeds that germinated and survived in all 15 plots of the indicated treatment versus date.

b. Percentage of seedlings first counted in May 1988 surviving versus date.

Many thistles that originally germinated in May disappeared during the summer. The rate of loss was greatest in the control plots and least in the clipped and dug plots. For the first month mortality rates were approximately equal in each treatment (15-20%), but between late June and late July mortality was much higher in the control plots than in the clipped plots or the clipped and dug plots (Fig. 2b). Between late July and late August the rate of loss in clipped plots was somewhat greater than in clipped and dug plots. By late August only 17.5% of the seedlings that had germinated in the control plots in May were still alive, while 31.5% of those on clipped plots and 48.6% of those on clipped and dug plots had survived.

Establishment of thistles was greatest in areas of intermediate soil moisture (Fig. 3). There was no establishment on sites in a meander scar which were covered with standing water for much of the spring nor on the driest sites, regardless of treatment. Observations in the field indicated that areas along the edges of meander scars and sloughs just above the level of standing water in the spring were most favorable for bull thistle establishment. The scatter in Figure 3 is partly due to differences between treatments; close examination reveals that at a given soil moisture level, clipped and dug plots generally had the most bull thistles and control plots the fewest.

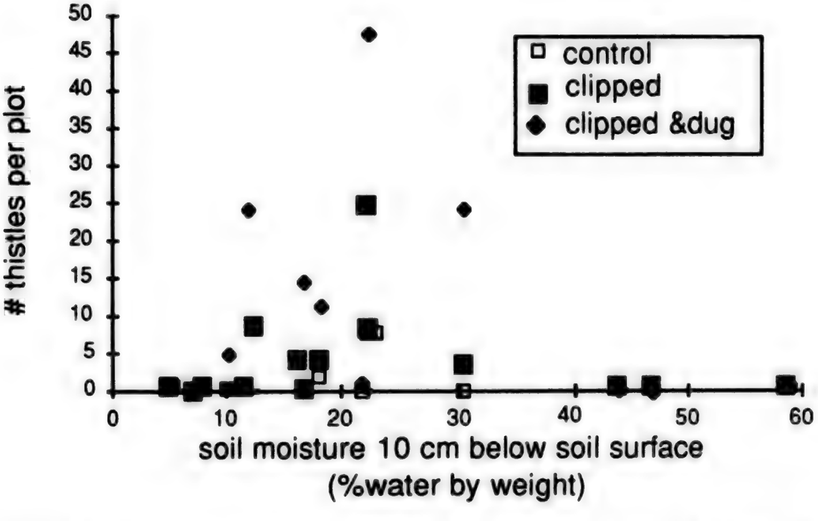

FIGURE 3. Disturbance experiment. Number of bull thistles per plot on 29 August 1988 versus soil moisture 10 cm below the surface in early July, 1988.

Manual Control Experiment

Over 80% of the adult thistles originally counted in the control plots were alive and flowering in September when their average density was 2.6 adults/m². Less than 2% of the adult thistles in the cut and remove plots resprouted, however, while slightly more than 5% of those in the cut and leave plots did. Mean density of adult thistles in the treated plots was just $0.25/m^2$ in September. Differences in survival between treatments were significant (ANOVA: $p < 0.0001$) and survival was significantly higher in control plots than in plots given either cutting treatment (Tukey's w procedure: 0.01). In addition, means of plant height and number of flowerheads produced were lower for plants that resprouted (44 cm and 3.7 flowerheads) than for adults in the control plots (85 cm and 15.8 flowerheads). Differences between cutting treatments were not significant, however, and date of cutting had no detectable influence on probability of resprouting.

DISCUSSION

The results of the disturbance experiment indicated that removal of vegetative cover promotes thistle establishment and that disturbance of the soil further promotes it. This agrees with Grimes' (1979) classification of bull thistle as a species of disturbed habitats and with results of studies in Australian pastures (Forcella and Wood, 1986; George et al., 1970) and Dutch coastal dunes (Klinkhamer and de Jong, 1988; van Leeuwen, 1981). Bull thistle establishment decreased exponentially with increasing cover of *Carex vesicaria* in a related greenhouse experiment (Randall, in press). The decrease was best described by the equation $y = 62.866\ \mathrm{EXP}[-0.064 * x]$, where y = percentage of thistle seeds that germinated and survived eight weeks and x = number of *Carex ramets* (tillers) per pot. The increase in seedling mortality with increasing cover was probably due to competition for light, since the pots were watered with a nutrient solution twice a day. Forcella and Wood (1986) hypothesized that grazing reduced competition from neighboring plants and thus increased chances that seedlings would become established. Klinkhamer and de Jong (1988) suggested changes in microtopography caused by soil disturbance may improve chances for soil-seed contact. Improved contact may encourage germination by increasing the number of microorganisms on the seed coat (van Leeuwen, 1981) and/or improving seed water balance. The low germination rates on control plots may likewise be explained by prevention of soil-seed contact by the thick litter layers present on most of them.

Seedling mortality was highest on control plots and lowest on clipped and dug plots in this study, and plots with low germination rates to begin with lost most of their seedlings (Fig. 2b). Van Leeuwen (1981) also reported lower mortality of bull thistle seedlings after soil disturbance, and suggested that differences in the microflora present on disturbed and undisturbed sites were responsible. Klinkhamer and de Jong (1988), however, found disturbance that increased seedling mortality. The timing of seedling mortality in the study indicated that differences between disturbed and control plots may have been due to increasing competition for water as the dry season progressed. Such competition would have been lowest in clipped and dug plots because the treatment killed roots of potential competitors. Potential competitors may have been handicapped to a lesser extent by the clipping treatment. The fact that all five plants that bolted were on clipped and dug plots also suggested resources were more readily available on such plots.

Klinkhamer et al. (1987) reported that some bull thistles that germinated in the fall or winter flowered by the end of the following summer. Similarly, musk thistle (*Carduus nutans*) may flower within one year of germinating, but only if it germinates in the fall or early winter (Lee and Hamrick, 1983). The plots in this study were checked at least monthly throughout the winter and spring; the five thistles that flowered in August were first noticed as seedlings in mid-May. It is possible that they were missed by earlier inspections, but not likely since they were on plots from which all other vegetation had been removed. In any case, it is remarkable that they grew to maturity so quickly. Klinkhamer et al. (1987) found that bull thistle rosettes must go through vernalization (a cold period) in order to bolt and flower. Interestingly, at the end of May, two weeks after seedlings were first noticed, a period of unseasonably cold weather with night-time temperatures below 0° C occurred in Yosemite Valley.

The disturbance experiment indicated that areas with "intermediate" levels of soil moisture are most favorable for thistle establishment. Seeds on the driest plots were probably unable to germinate due to lack of water. Many seeds were still visible on these sites in mid-summer. On the other hand, it is possible that seeds in the wettest plots in an old meander scar of the Merced River germinated but were unable to survive in standing water. De Jong and Klinkhamer (1988b) reported that bull thistle seedling survival was positively correlated with water content in the top 10 cm of the soil. They also found that artificial watering increased seedling survival and growth, and that year-to-year variation in rainfall affected seedling recruitment. Their study was conducted in a coastal-dune area with sandy soil and it appears that none of their sites were covered by standing water for extended periods.

The heaviest infestations of bull thistle in Yosemite Valley are often found on the banks of sloughs and old meander scars. These areas have relatively moist soil but are not under standing water for extended periods. The native vegetation in slough bottoms and on the lower portions of the banks is usually dominated by the hummock-forming species *Carex vesicaria* and *C. senta*. Large bare patches often exist between the hummocks in areas where water stands in the spring. These areas usually dry up as the season progresses, however, and the bare patches may then provide excellent sites for thistle establishment. Human footpaths frequently run along the upper edges of the banks creating more openings for thistles. Other thistle infestations in Yosemite Valley are in areas with obvious gopher activity. Gophers and/or other small burrowing animals eat thistle roots but do not seem to attack many plants in a given area. A type of "positive feedback" between gopher activity and thistle establishment may exist, in which gophers continue to dig in an area over a period of years, keeping it in a disturbed state conducive to thistle establishment in part because the thistles are a food resource for them. The importance of gopher activities for the establishment of ruderal plants in a California grassland was documented by Hobbs et al. (1988).

Cutting adult thistles at the soil surface in an effort to kill them proved to be surprisingly effective. Studies in pastures indicated that mowing effectively controlled infestations of bull thistle (Harris and Wilkinson, 1984) and the related musk thistle (McCarty and Hattling, 1975). Observations by people who had participated in efforts to control bull thistle in Yosemite Valley, however, indicated that they resprout. The ability of adults to resprout after cutting was of primary interest in this study, but bull thistles may bolt any time between mid-June and late August in Yosemite Valley; therefore, rosettes in treatment plots were cut to prevent them from bolting later and being confused with resprouted adults. Rosettes are usually far more numerous than adults in a given area, however (over 200/m^2 in densely infested locations), and attempts to cut them would be extremely inefficient and probably lead to extensive damage of the remaining vegetation. Thus, although there was no detectable difference in the probability of resprouting due to date of cutting, it would be most efficient to cut late in the season when most of the plants have bolted but before significant numbers of them flower. Observations in a nearby meadow, where adults were removed from a two-hectare area by a California Conservation Corps crew (to be published elsewhere), indicated that plants cut 20 cm or more above the surface have a greater chance of resprouting. A second sweep through these areas approximately one month after the first could probably be done rapidly and result in the elimination of most flowering plants.

Plants that did resprout were on average shorter and produced fewer flowerheads and presumably fewer seeds than adults that survived in control plots. Klinkhamer et al. (1988) noted that variations in seed production have more influence on population fluctuations of biennials than do variations in adult mortality, and Klinkhamer and de Jong (1988) found that seed input limited bull thistle establishment on coastal dunes in the Netherlands. Thus, although some plants resprout, cutting thistles may help control their populations by limiting seed production. In addition, because the travel distance of wind-dispersed seeds is positively correlated with the height at which they are released (Harper, 1977; Sheldon and Burrows, 1973), the decrease in plant height may limit the spread of those that are produced.

Efforts to control thistle populations manually would be greatly complicated if a large reserve of seeds persisted in the soil from year to year. It is not known whether a significant seed bank exists in Yosemite Valley, but little evidence of one was found in coastal populations in the Netherlands (Klinkhamer and de Jong, 1988; Klinkhamer et al., 1988; van Breeman and van Leeuwen, 1983) or in British populations (Roberts and Chancellor, 1979). A reserve of rosettes will remain after the adults have been killed, however, and an individual may remain in the rosette stage for up to five years before flowering (de Jong and Klinkhamer, 1988a; Klinkhamer et al., 1987). It may be possible to control populations in a given area if the adults are killed before they set seed, but only if the program is continued for at least four years.

The number of thistles resprouting was slightly greater in cut and leave plots than in cut and remove plots, but the difference was not significant and may have been the result of plants being missed because they were covered by recently cut neighbors. It is likely that a dedicated work crew clearing an area several hectares in extent would miss living thistles covered by recently cut neighbors more frequently than a researcher carefully clearing 2 x 2 m plots to determine treatment differences. Seeds may be produced and released from bull thistle stems cut 5-10 days after their flowers open (personal observation), and McCarty and Hattling (1975) found that seeds matured on the closely related plant, musk thistle, when it was cut just four days after its flowers opened. Thus, apart from aesthetic reasons, it is advisable to remove the plants from an area after they have been cut. This can create another problem if the area is heavily infested; how can the plants be disposed of? Thistles from the two-hectare area cleared by the California Conservation Corps crew filled seven dumpsters. If thistles are to be removed from a large area, plans for disposing of them should be made beforehand.

CONCLUSIONS

1. Disturbance encouraged germination and establishment of *Cirsium vulgare*.
    a. *Cirsium vulgare* establishment was promoted by removal of vegetative cover and further promoted by disturbance of the soil.
    b. A few individuals did invade undisturbed areas with 100% vegetative cover.
2. Success of *Cirsium vulgare* was greatest in areas of "intermediate" soil moisture, such as along slough banks.
    a. Germination rates were extremely low in plots with the driest soil.
    b. Thistles did not become established in wet, low-lying areas subject to long periods under standing water.

3. Cutting adult *Cirsium vulgare* at the soil surface may be an effective method of manual control.
   a. The date of cutting had no detectable influence on probability of resprouting.
   b. Removing cut thistles from the area decreased the probability of resprouting slightly, but the difference was not significant.

MANAGEMENT ALTERNATIVES

The management alternatives offered below are based on just one year of data from a single area and should be regarded as preliminary.

1. Removal of vegetative cover and disturbance of the soil promotes bull thistle establishment. Therefore, reduction or elimination of human-caused disturbances, such as footpaths in or near infested areas, is advisable.
2. Manual removal of adult bull thistles may be an effective way of controlling their populations. This work indicated that the plant's root need not be pulled or grubbed out of the soil to kill it. They may be cut at or near the soil surface instead. A machete was excellent for this purpose. It was easiest to pull the smaller adult plants and cut those that were larger, especially if they branched at or below the soil surface. Pulling larger plants will likely disturb the soil, providing a seedbed for thistles or other ruderal species. Removing plants that have already been cut makes it easier to see those that remain. Another reason to remove them is that they may still form seed if cut shortly after flowering. Before a cutting program is begun, however, plans should be made for disposing of the plants.
3. It would be most efficient to cut late in the season when most plants have bolted but not yet flowered. A second sweep through these areas approximately one month after the first might be done rapidly and may result in the elimination of most flowering plants. A reserve of rosettes will remain after the adults have been killed and individuals may remain in the rosette stage for up to five years. Thus, cutting may need to be continued for four years or more in order to control populations in a given area. After this time, efforts may still be required to eliminate thistles that germinate from dormant seed or seed arriving from distant locations.
4. Preliminary observations indicated that bull thistles cut 10 to 40 cm above the soil surface are more likely to resprout than those cut at the soil surface. It is not known whether these resprouted plants are capable of producing significant amounts of seed. Experiments to test the effects of different cutting heights are planned for the 1989 growing season. Results of this work will be made available.

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Post-grazing Grassland Succession in Golden Gate National Recreation Area

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Abstract. Cattle were removed from the Tennessee Valley area of the Golden Gate National Recreation Area in 1984. Between 1985 and 1988, cover of forbs and non-native annual grasses decreased on valley and hillside plots representing two soil types. Two alien forbs, Italian thistle (*Carduus pycnocephalus*) and poison hemlock (*Conium maculatum*), increased in the areas. Native perennial grass cover increased where grazing was removed. On the hillside plots, cover of native perennial grasses increased through 1987 and then declined in 1988. The hillside areas succeed from forbs to annual grasses as native grasses slowly increase in cover. Valley area succession fluctuated with different species of annual grasses dominating each year.

INTRODUCTION

The Marin County landscape has been shaped by grazing. Native tule elk (*Cervus canadensis nannodes*) grazed the land prior to the introduction of cattle. The prairie was characterized primarily by native perennial bunchgrasses (Bartolome and Gemmill, 1981). Europeans introduced Mediterranean annual grasses and herbs along with cattle. Subsequently, the present-day California native species diminished and the California annual grassland developed.

Grazing was removed from the Gerbode Valley in Golden Gate National Recreation Area (GGNRA) in the early 1970s. An influx of alien thistles (*Carduus pycnocephalus, Silybum Marianum*) and poison hemlock (*Conium maculatum*) followed. These showy aliens are still dominant along drainages and in 1-5 acre patches throughout the Gerbode Valley.

Cattle were removed from Tennessee Valley, one valley north of the Gerbode Valley, in the fall of 1984. A monitoring system was undertaken to document the post-grazing succession of the grassland and learn more about alien plant invasions. Two management objectives of the National Park Service in these areas are to restore the California perennial coastal prairie and to reduce alien plant invasion. Establishing a vegetation monitoring program was viewed as an important first step.

STUDY AREA

The Tennessee Valley area is located in southern Marin County, California. The area contains two major soil types. The Rodeo Clay Loam soil type is found in the valley close to Elk Creek at an elevation of 30 feet above sea level. This deep, poorly drained soil formed in mixed alluvium derived from chert, sandstone and granite. Permeability of this Rodeo soil is slow. Available water capacity is very high (Kashiwagi, 1985). This fine-grained soil supports mainly annual grass and forb vegetation. Alien plants readily invaded this soil type in the Gerbode Valley when grazing was removed. The second major soil type is the Tamalpais-Barnabe Variant; a gravely loam found on the hillside areas of Tennessee Valley at elevations of about 40 to 800 feet. Permeability of the Tamalpais soil is moderately slow and available water capacity is low. The Barnabe Variant soil is shallow and well drained. Both variants are formed in material derived from chert or sandstone (Kashiwagi, 1985). The vegetation is mainly brush, annual grasses and forbs. Alien plants, mainly thistles, invaded this soil type in the Gerbode Valley when grazing was removed. The hillside plots have a slope of 22-25% and a southwest aspect while the valley plots are level.

Both soil types have been grazed in the Tennessee Valley area throughout recent history. Since the establishment of Golden Gate National Recreation Area in 1972, grazing was permitted during the months from March to October. Grazing ceased on 4 September 1984.

METHODS

Residual dry matter (RDM) was collected from 50 randomly located circular plots (0.96 ft^2) in each of the two study units. Each plot was hand clipped to ground level on the same day the cattle were removed, September 4, 1984. The RDM was then dried and weighed. RDM was again collected in 1985 to assess changes after the removal of cattle.

Five 20-meter transects and photopoints were established in each study site in April, 1985. Transects were established in a stratified random manner in chosen representative areas. An initial bearing was randomly chosen within the representative area. Five transects were systematically placed perpendicular to the random bearing at two-meter intervals. Along each transect, ten 0.96-ft^2 circular plots were placed every two meters beginning at 0.5 m.

The Domin-Krajina index (Mueller-Dombois and Ellenberg, 1974) was used to estimate cover in each of the 50 plots for each group of annual grasses, native grasses and forbs. The Domin-Krajina scale has a finer index than most cover indices and allows separation of low density units. In this study, the scale was defined as follows:

+ solitary with insignificant cover
1 1 or 2 plants
2 cover less than 1%
3 1-5% cover
4 6-10% cover
5 11-25% cover
6 26-33% cover
7 34-50% cover
8 50-75% cover
9 greater than 75% cover (75-98%)
10 approximately 100% cover (98-100%)

For tabulation, the indices were converted to the midpoint value of the range. All species present, in each plot, were identified at the genus level.

Sampling dates for both the hillside and valley plots were 29 April 1985, 29 April 1986, 21 April 1987 and 20 April 1988.

RESULTS

Residual dry matter (RDM) was 990 lbs/acre when the cattle were removed from Tennessee Valley in 1984. RDM increased to 1700 lbs/acre in 1985.

Hillside Plots

On 29 April 1985, 69.0% forb cover was observed. Italian thistle (*Carduus pycnocephalus*) comprised only 0.01% of this (Table 1). The most abundant forbs were lupine (*Lupinus bicolor*), California poppy (*Eschscholzia californica*), plantain (Plantago sp.), geranium (*Geranium* sp.), *Microseris* sp. and *Viola* sp.

Alien annual grasses, representing over 28% cover, included *Bromus mollis*, *Avena barbata*, *Lolium multiflorum* and *Vulpia* sp. Native perennial grasses (*Stipa pulchra* and *Danthonia californica*) comprised only 1.6% cover (Table 1).

In 1986, forb cover (mostly plantain and poppy) had decreased to 40% with a marked disappearance of lupine. Annual grass cover, dominated by the same species as seen the previous year, increased. Perennial grass cover showed a slight but statistically insignificant increase (Table 1).

By 1987, forb cover had further decreased but geranium and Italian thistle became the dominant species along with poppy and lupine to a lesser extent.

Cover of native perennial grasses increased to a high of 11.1% in 1987 but decreased to 5.2% in 1988. All vegetation groups showed decreases in cover in 1988 while Italian thistle increased from previous years.

TABLE 1. Mean percent cover (standard error in parentheses) of hillside plots from 1985 to 1988 and number of the 50 plots containing Italian thistle.

	COVER				
Year	Exotic Annuals	Forbs	Native Perennials	Total	# Plots W/Thistle
1985	28.1	69.0	1.6	98.2	14
	(3.9)	(3.9)	(1.3)		
1986	38.7	40.1	2.9	82.0	12
	(4.2)	(4.4)	(1.8)		
1987	20.2	26.7	11.1	55.2	11
	(3.0)	(3.6)	(3.6)		
1988	17.0	21.6	5.2	43.8	19
	(2.5)	(2.7)	(2.0)		

TABLE 2. Mean percent cover (standard error in parentheses) of valley plots for 1985 to 1988 and number of the 50 plots containing Italian thistle.

	COVER				
Year	Exotic Annuals	Forbs	Native Perennials	Total	# Plots W/Thistle
1985	94.9	13.4	0.0	106.8	0
	(2.1)	(2.3)	(0.0)		
1986	95.0	2.9	0.0	98.8	2
	(2.6)	(2.6)	(0.0)		
1987	56.2	9.9	0.0	65.6	9
	(4.1)	(3.3)	(0.0)		
1988	62.8	21.6	5.2	64.4	3
	(2.2)	(2.0)	(0.8)		

Valley Plots

In contrast to the hillside area, the valley plots were dominated by annual grasses (mainly *Hordeum leporinum* and *Lolium multiflorum*) in 1985 (Table 2). Forbs represented only 13.4% of the cover, with the majority being a clover (*Trifolium* sp.). Geraniums and composites were also present. No native grasses were present.

In 1986, one year following removal of cattle, Mediterranean annual grasses continued to dominate the site, although a greater species mix was noticeable, including *Bromus mollis*, *B. rigidus* and the native perennial grass *Hordeum brachyantherum*. Forb cover decreased to 2.9% primarily because clover disappeared completely. Poison hemlock invaded the area from the stream side (west side) and was the dominant forb. Plantain (*Plantago* sp.), morning glory (*Convovulous* sp.), dock (*Rumex* sp.) and meadow rue (*Thalictrum polycarpum*) were also present.

In 1987, Mediterranean annual grasses still dominated the area (Table 2). Annual grass species included *Hordeum* near the stream, *Lolium* and *Bromus* in the center, moving east, and *Lolium*, *Vulpia* and *Bromus* the furthest east. Native species included *Hordeum brachyantherum* and one perennial rush (*Juncus patens*). Poison hemlock dominated the forb cover. Seven percent of the 9.9% (a relative 76%) cover of forbs was poison hemlock.

In 1988, valley plots showed an increase in alien annual grasses, especially *Lolium multiflorum*, the dominant species. Forb cover greatly decreased to 1.7% as poison hemlock cover decreased to near zero in 1988.

DISCUSSION

Total vegetation cover showed a continuous decrease for the three years following the removal of grazing (Tables 1 and 2). The largest single year decrease occurred in 1987. Two successive years of low rainfall may have been responsible for the large decrease in cover; however, other possible contributing factors include the increase in residual dry matter and decrease in fertilizer due to cattle removal. RDM affects germination, generally resulting in less forb seed germinating than grass seed (Peart and Foin, 1985). The winter of 1987 had negligible rainfall in November and December. 1988 was also a low rainfall year; even though the precipitation in November and December was good, January and February were below normal. The alien annual grasses increased the first year after grazing was removed and then decreased. The low rainfall of 1987 may be responsible. Annual grass germination responds to the rainfall during the year (Raguse et al., 1977).

Forb cover on the hillside plots has continually decreased since 1985. This is typical of what is often seen when grazing is removed from other areas (Foin and Hektner, 1986). Tennessee Valley had a colorful display of poppies and lupines during the first year after the cattle were removed. This display has greatly diminished since that time.

The valley plots had a similar decrease in forb cover. The increase in 1987 was mainly due to an influx of one species, poison hemlock. The influx is interesting since it occurred in a low rainfall year. Poison hemlock is most common on shady or moist ground (Parsons, 1973) and could be expected to show a decrease during dry years. This population did show a decrease after two low rainfall years (1987-88). Native perennial grasses greatly increased on the hillside plots. This occurred despite the low rainfall and may reflect their ability to compete with the annuals under stressful conditions. Cover decreased after two low rainfall years, but still represented a 383% relative increase above the 1985 level.

CONCLUSIONS

Although longer-term studies are needed, preliminary results indicate the following:

1. Native perennial grass cover has increased on these soil types where grazing has been removed;
2. Early post-grazing succession on Rodeo Clay loam soils, with little slope or aspect, fluctuates annually and is complex in nature;
3. Post-grazing on Tamalpais-Barnabe Variant soil, with 20-30% slope and a southwest aspect, begins with forb cover dominating and then changes to annual grasses dominating as native grasses slowly increase in cover;
4. Problem alien forb species on Rodeo Clay loam soils with little slope or aspect include poison hemlock, Italian thistle and milk thistle; and
5. The principal problem alien forb species on the Tamalpais-Barnabe soil variant (with 20-30% slope and a southwest aspect) is Italian thistle.

MANAGEMENT ALTERNATIVES

The two management objectives addressed by the monitoring study are (1) to restore the California perennial coastal prairie, and (2) reduce exotic plant invasion. These objectives are supported by National Park Service policies addressing natural area maintenance of native species and the reduction of exotic species (USDI, 1989).

Promoting native perennial grasses in the presence of alien forbs and grasses is a topic of interest to many land managers. Possible management alternatives include the following: (1) do not interrupt present successional trends; (2) introduce grazing at specific times of the year to enhance native perennials and deter non-native species; (3) mow fields at selective times of the year; (4) prescribe-burn at selective times of the year; (5) plant or seed native perennial grasses; (6) use selective herbicides on non-native species; (7) hand-weed non-native species; and (8) use biological control techniques.

Two of these techniques, herbicidal and biological control, will not be addressed in this paper. The National Park Service policy on non-native species is to explore all other methods prior to considering herbicides; and therefore, it is not considered in this paper. Biological control methods are known for Italian thistle; however, to my knowledge, none are known that will not attack native thistles as well. No known, approved biological control exists for poison hemlock or specific non-native annual grasses.

Mowing may be effective in reducing Italian thistle; however, the ability to mow is restricted by topography and often by time of year. Mowing is most effective at flowering time (May-June), before the seed heads are produced (J. McHenry, pers. comm.; Parsons, 1973). However, plants can regrow from the base and plants cut close to flowering time can produce seed on the cut portion of Italian thistle (Tasmanian Department of Agriculture, 1977). One mowing will not be effective and several years of treatment should be anticipated. Mowing, prescribed fire and grazing defoliate all plants on the site. Therefore, effects of fire and grazing on native perennial grasses, as well as alien species, must be considered. Little research has been done on fire in the native perennial grasslands of California, although fire is recognized as an important component in the maintenance of grasslands. The high density of dominant annual species in these grasslands makes it difficult to extrapolate other research to the perennial grassland mix in California (Menke, 1989).

Dennis (1989) provided valuable information concerning defoliation of four native perennial grasses (*Poa scabrella, Melica californica, Koeleria cristata, Stipa pulchra*) present within an annual grassland. Her clipping studies concluded that *Stipa pulchra* has a different response to clipping than the other native species. Therefore, many of the generalizations made from *Stipa pulchra* studies about native grasses in general are erroneous. Clipping at times that promoted *Stipa pulchra* was detrimental to *Melica californica* and *Poa scabrella*. Clipping following senescence was the only method that had potential for enhancing the growth of these and perhaps other perennial species.

The presence of non-native annual grasses negatively affects the growth of the native perennial grasses (Dennis, 1989). However, experiments on established native perennial grasses and *Bromus mollis* seeding concluded that the presence of the annual bromus had little effect on the native perennials. Dennis (1989) concluded that a well established stand of native perennial grasses, under some circumstances, may be capable of resisting reinvasion by annual grasses even on sites suitable for lush growth of annuals.

One management implication of this information is that prescribed fire, mowing and grazing techniques must be carefully planned with the ecology of desired species in mind. Certain management practices may discourage some natives while encouraging others.

A second implication is that planting native perennial plants and establishing a perennial presence may succeed in retaining the native composition once it becomes established, thus preventing reinvasion by non-native annual grasses.

Little information could be found regarding control of poison hemlock. The staff at John Muir National Historic Site found that repeated mowing (three times in one season) was effective. However, this treatment is difficult in the steep drainages of GGNRA where the species exists. Experiments with cutting and pulling programs are encouraged in small areas where control appears possible with little effort.

The complexity of managing for a native perennial grassland requires a careful application of techniques. Since the native perennial grasses are successfully competing with the annual grasses thus far, a non-aggressive management scheme is recommended. A clipping and mowing experiment with both Italian thistle and poison hemlock would help to determine the potential success of those techniques in reducing alien forb numbers.

Propagation and planting of native grass plugs into the annual grassland is recommended on a continual basis in order to create a dominant perennial stand that may deter annual grass domination. Plantings of *Stipa pulchra*, *Danthonia californica*, and *Elymus triticoides*, for example, have been successful in California state park projects (J. Barry, pers. comm.).

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SECTION III: PHYSICAL PROCESSES

Introduction to the Section on Physical Processes

In addition to the inventory and monitoring of fauna and flora, the National Park Service has a responsibility to manage natural resources "with a concern for fundamental ecological processes as well as for individual species and features" (USDI 1988, Management Policies 5:1). Research must be geared to detect and accurately predict changes in resource conditions resulting from natural ecological processes and must consider how those processes may have been altered by human activities. Resource managers must be trained to evaluate the magnitude and direction of those changes attributable to human activities, and make management decisions to maintain the closest approximation of the natural ecosystem. Physical and biological processes are intimately related. Understanding the role of fire, erosion, litter accumulation and decomposition, and other processes requires information on the interactions of multiple species with abiotic resources and physical processes on a watershed, landscape or global scale. As an agency and a society, we are only in the earliest stages of developing that understanding.

Francisco L. Perez shows that high elevation tree islands in Lassen Volcanic National Park play an important role in alpine soil evolution. Aside from critical information on fuel accumulation rates (important to fire managers), we gain an appreciation of slow soil evolution processes, such as changes in soil bulk density and organic matter accumulation as influenced by tree species composition and elevation. Perez's data might also be used to assess changes in the productivity of timberline tree species as they respond to global climatic change; these data can subsequently be appplied to predict production in northerly or high elevation trees (EPA, 1988).

The fact that ecological processes often operate on landscape scales is illustrated in James H. Popenoe's research on the landslide susceptibility of soils in Redwood National Park. The author points out the importance of soils as a natural resource but notes that among federally administered lands in California, the national parks are conspicuous for their lack of standardized soil surveys (USDA, 1988). In Redwood National Park, however, soil survey information is the primary tool for predicting management needs for watershed rehabilitation following pre-park deforestation. The information is standardized according to the format of the National Cooperative Soil Survey, thus making it consistent with soil information gathered by other federal and state agencies.

Fire is another ecological process that impacts many ecosystem components and ignores political boundaries. The last three papers in this volume touch upon the ecological, political and social aspects of fire. Thomas J. Stohlgren proposes vegetation monitoring guidelines for chaparral in Sequoia and Kings Canyon National Parks, which at first glance appear site-specific. However, his approach represents a logical, broad-based lesson in proper experimental design, applicable for sampling vegetation and fire effects monitoring in many other vegetation types and areas. The key steps in developing a long-term monitoring program, according to Stohlgren, are: (1) articulating management objectives; (2) reviewing the scientific literature for appropriate sampling techniques; (3) deciding which ecosystem parameters to measure or monitor; (4) determining the level of precision required of the sampling technique to meet management objectives (i.e., testing for appropriate plot and sample sizes); and (5) providing a framework for evaluating results. Perhaps symptomatic of many vegetation studies (see Hurlbert, 1984), including ongoing monitoring efforts in many national parks, too few examples come to mind where proper experimental designs have been employed in the past. Closer public scrutiny in the future may help ensure that fire effects monitoring is well-planned, accurate and repeatable.

In addition, the aftermath of the Yellowstone National Park fires of 1988 found the National Park Service and the public more aware of the political constraints of fire management. In their paper, Little and Schonewald-Cox examine the role of administrative boundaries and jurisdictional authorities in the management of fire in and around Lassen Volcanic National Park. They use the 1987 Snag Fire as an example of how interagency fire control responses did not necessarily meet the park's fire management objectives. In fact, because of the perceived danger of fire spreading onto land under Lassen National Forest (U.S. Forest Service) jurisdiction, the authors estimate that only 39% of the park habitat is under the sole fire management control of Lassen Volcanic National Park. Increased communication, cooperation and compromise are needed among land management agencies. Our natural resource conservation strategies must extend beyond our boundaries.

The final paper by David J. Parsons addresses several aspects of fire, including policy, public involvement and the role of science in the management of fire in natural ecosystems. He describes the development of a comprehensive research program to better understand the potential of restoring fire as a natural process in giant sequoia (*Sequoiadendron gigan-*

teum) groves in Sequoia, Kings Canyon and Yosemite National Parks. One key component was a review of the current fire management program by an independent panel of outside experts. The panel's recommendations included the need for an extensive research program on fire history, demographic and life history studies, fuel dynamics, computer simulation models, visitor response and fire effects. Parsons again points out that, for most ecosystems, we lack a detailed understanding of basic ecosystem structure and function. While the National Park Service was able to fund a comprehensive study of the role of fire in giant sequoia, overall agency research funding has been limited and many National Park Service managers must continue to manage natural resources without adequate information.

That is the challenge before us today. We must remember that the primary goal of natural resources management is to "manage the natural resources of the National Park System to maintain and perpetuate their inherent integrity" (USDI, 1988). Our efforts to inventory and monitor the flora, fauna and physical processes of the parks must be designed to achieve this goal.

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Conifer Litter and Organic Matter Accumulation at Timberline, Lassen Peak

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Abstract. Tree size, plant litter, and organic matter accumulation were studied in 62 tree islands of whitebark pine (*Pinus albicaulis*) and mountain hemlock (*Tsuga mertensiana*) at timberline in Lassen Volcanic National Park, between 2550 and 3060 m altitude. Tree height decreased with altitude; high-elevation trees were prostrate. Litter depth decreased away from tree stems, from an average of about 60 mm next to them to less than 4 mm at 3 m distance. Litter accumulation beneath trees decreased with increasing elevation, from greater than 30 to less than 1 kg/m^2; pine islands had higher litter accumulation than hemlock ones found at the same altitude. Organic matter content of the soil was influenced by litter deposition. Soils below pines had higher organic matter percentages; control samples collected away from tree islands showed very low organic matter levels. Tree islands play an important role in alpine soil evolution through the addition of litter to the soil surface. Some soil properties, such as bulk density, are highly dependent on organic matter, thus considerable differences were found when tree-island soils were compared with bare control soils.

INTRODUCTION

The quantitative study of the litter layer accumulated beneath trees appears to have been initiated by Lowdermilk (1930) in California, where he collected litter from different forest types. Some of the studies that followed, which called this litter layer the "forest floor," sampled forest litter in the vicinity of Lassen Volcanic National Park (Bodman, 1935; Bodman and Kittredge, 1938; Dickson and Crocker, 1953) as well as in other California forests (Kittredge, 1940, 1955; Zinke and Crocker, 1962). These studies were restricted to low-elevation forests, and concentrated on comparing litter accumulations among a variety of forest types, but did not investigate the local patterns of variability of litter accumulation within a given forest.

In addition to the study of forest floors, annual litterfall has been investigated in western forests (Tarrant et al., 1951; Stohlgren, 1988). However, annual litter production usually shows a pronounced interannual variability (Tarrant et al., 1951; Bray and Gorham, 1964; Challinor, 1968; Millar, 1974; Stohlgren, 1988); thus, long sampling periods are needed in order to obtain "average" annual litterfall values.

Several recent publications have compiled data on plant litter production and decomposition rates along latitudinal gradients (Bray and Gorham, 1964; Jordan and Murphy, 1978; Meentemeyer et al., 1982; Meentemeyer, 1984). These have shown that litter production generally decreases with increasing latitude, primarily in association with the reduction of annual average temperatures and available solar energy during the growing season. Since decomposition rates are positively correlated with temperature (Williams and Gray, 1974; Meentemeyer, 1984), litter breakdown decreases at higher latitudes. A general similarity in litter production trends along latitudinal and altitudinal gradients should be expected, since annual average temperatures and the length of the growing season also decrease at higher elevations (Millar, 1974). As a result, plant productivity drops with altitude (Arno, 1984); therefore litterfall should also decline. However, since litter production depends on a number of factors (Walsh and Voigt, 1977), the pattern is not always clear. Ebermayer (1876) and Reiners and Lang (1987) studied conifer litterfall along altitudinal gradients. They found an upwards increase of litterfall at lower elevations, followed by a rapid decline in litter production at higher altitudes.

Litter decay rates also decrease with elevation due to the climatic gradients previously mentioned (Shanks and Olson, 1961; Williams and Gray, 1974; Edmonds, 1980). The combined effect of these trends of decreasing litter productivity and decomposition rates with altitude is not clearly discerned from the literature. Slower decomposition rates should lead to "large accumulations of litter and debris on the forest floor" of subalpine forests (MacMahon and Andersen, 1982, p. 375). Yet, other authors (e.g., Rundel et al., 1977, p. 579) indicate that in the subalpine forests of California, "litter and fuel accumulations are low." Litter dynamics are further complicated by the fact that, in many cold temperate forests, belowground inputs of organic matter from fine root turnover can be substantially larger than those of aboveground litterfall (Grier et al., 1981; Vogt et al., 1986).

The significant role of plants in soil formation through the addition and incorporation of organic debris into the mineral soil has been stressed by many authors (Dickson and Crocker, 1953, 1954; Zinke, 1962; Zinke and Crocker, 1962; Perez 1987), but few studies have compared the effectiveness of different plant species in altering soil properties (cf. Mergen and Malcolm, 1955; Zinke, 1962; Challinor, 1968; Page, 1974; Walsh and Voigt, 1977). These studies show that the amount and type of litter-derived organic matter incorporated into the soil significantly influences several soil properties. Soil bulk density is rapidly lowered with increasing organic matter (Dickson and Crocker, 1954; Zinke, 1962; Zinke and Crocker, 1962; Page, 1974). The effect of organic matter on bulk density is so pronounced that this is often estimated from organic matter content alone (Saini, 1966).

RESEARCH OBJECTIVES

This study reports on some of the characteristics of isolated trees (tree islands) found at timberline in Lassen Volcanic National Park, California. The main goals are to:

1. Ascertain the nature of the spatial patterns of litter accumulation beneath timberline trees along an altitudinal gradient from 2550 to about 3060 m elevation.
2. Compare the ability of two timberline tree species found in the study area, whitebark pine (*Pinus albicaulis* Engelm.) and mountain hemlock (*Tsuga mertensiana* (Bong.) Carr.), to accumulate plant litter.
3. Determine the influence of the litter layer on the organic matter content of soils within tree islands of these two timberline tree species.
4. Determine the effects of organic matter buildup on soil bulk density.

STUDY AREA

Lassen Volcanic National Park is in northern California (Fig. 1A). The highest point in the park is Lassen Peak, an isolated dome volcano 3193 m high which marks the southernmost point of the Cascade Range, at 40° 29′ N and 121° 30′ W. Lassen Peak has a Mediterranean climate, with precipitation concentrated in the October-April period. Annual total precipitation ranges from 1300 to 1700 mm. Most of the precipitation falls as snow (Major, 1977), which is re-distributed by the prevailing westerly winds.

Two tree species, whitebark pine (*Pinus albicaulis*) and mountain hemlock (*Tsuga mertensiana*), are found at timberline; beyond the forest line—general upper limit of continuous forest (Arno, 1984)—at 2500-2600 m, tree islands of both species extend up to about 3060 m (Figs. 1B and 2). Soils above the forest line have scarcely been studied; they are sandy-skeletal Lithic and Typic Cryorthents with a high gravel content—about 60%. The soil fraction is dominated by sand (77-93%); silt is only 6-21%, while clay content is very low, 0.3-2.0% (Perez, 1989). This particle size distribution suggests that frost weathering is the primary agent of soil formation above timberline (Perez, 1984).

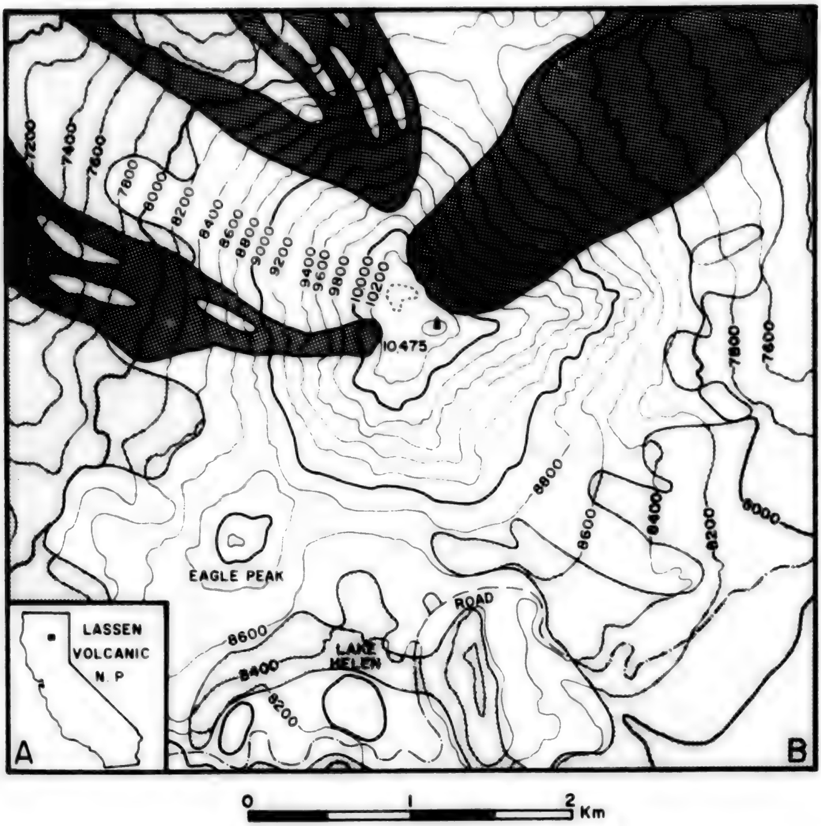


FIGURE 1. (A) Location of Lassen Volcanic National Park. (B) Lassen Peak area. Dark shading shows areas recently affected by volcanic activity (adapted from Kane 1980). Light shading shows the approximate extent of subalpine forest. Base maps: Lassen Peak, Manzanita Lake, and Mt. Harkness (1956), Prospect Peak (1957). U.S. Geol. Surv., 15′ quadrangles, 1:62,500. Elevations in feet.

METHODS

Field work was conducted during the summers of 1982 and 1983. Thirty-three whitebark pine and 29 mountain hemlock tree islands were sampled between 2550 and 3060 m. Only trees on the southern flank of Lassen were sampled, because the W, N, and NE slopes have been disturbed by recent volcanic activity (Fig. 1B). Tree islands studied had predominantly a SW or SE exposure (Fig. 3). Elevation was measured with an altimeter—estimated error: ± 10 m; slope aspect was measured with a compass. Tree height was measured with a clinometer and tape.

Litter depth was measured (cf. Kittredge, 1940) at 0, 1, 2, and 3 m distances from the tree stem along a transect extending downslope. In a few

FIGURE 2. Southern flank of Lassen Peak, showing the upper edge of the subalpine forest. Lake Helen is in the foreground. Photograph taken August 1982.

cases, due to inaccessibility, the transect extended sideways. Litter was sampled at 1 m distance by cutting a 15 × 15 cm square with a sharp knife (cf. Kittredge, 1955); large stems or twigs on the surface were not included. All samples were located within the tree crowns; a few samples below trees with smaller crowns had to be collected 50 cm away from the stem. Litter compaction was measured with a pocket penetrometer; five replicate measurements were taken at each site. The mineral soil beneath the litter was sampled with a 120-cm^3 iron cylinder, which was gently pressed into the upper 8 cm of the soil so as not to disturb its field bulk density; 35 additional bare soil samples were collected near tree islands. These will henceforth be called alpine soils.

Plant litter was air-dried inside paper bags for two months, and carefully cleaned of mineral particles, then oven-dried at 100° C for 24 hours (Kittredge, 1955) and weighed. Amounts of dry litter are given in kg/m^2. Soil samples were oven-dried and weighed to determine field bulk density. Organic matter content of soil samples was calculated with their loss of weight on ignition.

The frequency distributions of litter depth were not normally distributed, thus values were transformed to Log_{10}; subsequently, data sets tested as normal (p = 0.01) (Jones, 1969). Geometric mean litter depth values are reported. The relationships between individual variables were analyzed with linear regressions. In most cases, data transformed to Log_{10} gave better correlations than non-transformed values; therefore, log-transformed data were used. Except where noted, the correlation coefficients (r) are given at the p = 0.001 significance level.

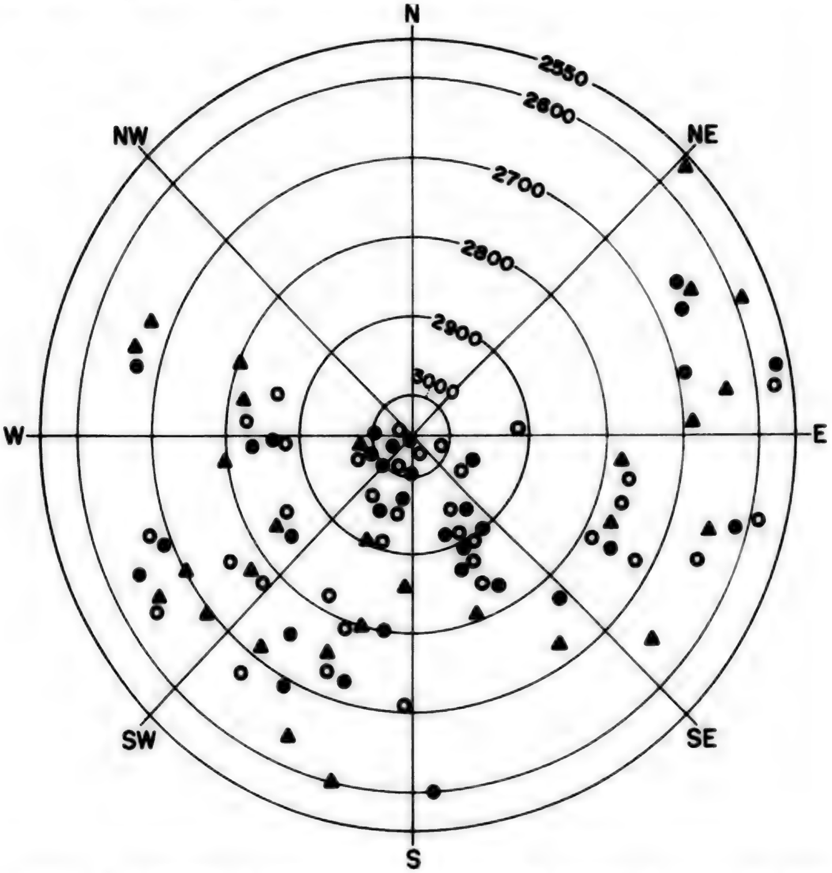


FIGURE 3. Distribution of samples by altitude (circles and triangles, in meters) and orientation. Key: dark circles = whitebark pine, dark triangles = mountain hemlock, light circles = alpine soil samples.

RESULTS

The Timberline Zone

Mountain hemlock forests extend up to 2600 m. This level coincides with the base of the talus slopes. Water seeps through the coarse debris and appears at the talus foot, providing the high moisture levels needed by mountain hemlock. Probably for similar reasons, hemlock islands dominate the lower timberline, while the upper timberline is characterized by groups of pine (Fig. 4). The highest hemlock located was at 2985 m, while the highest pine was found at 3060 m. Both species attain their maximum elevation in the SW and W flanks (see Fig. 3).

Hemlock islands are commonly composed of several trees; in some areas, trees form strips perpendicular to the slope and to the wind direction (Fig. 5A). These ribbon forests (Arno, 1984) develop mostly along rocky ridges and are associated with snowdrifts deposited downwind from them; these prevent further seedling colonization and island expansion. Pines can also form extensive groups, but are commonly found as isolated individuals on rocky spines extending parallel to the slopes (Fig. 5B). Pines also grow directly on the talus, but here are found downslope from large boulders or small outcrops; these locations are more stable than the surrounding slope (Perez, unpubl. data; cf. Perez, 1988).

There was a gradation of growth-forms of both tree species along the altitudinal transect, from erect single-stem trees, to erect multi-stemmed trees, to elfinwood trees growing prostrate (cf. Clausen, 1965). This trend was most pronounced in the pines (Fig. 6), which displayed only elfinwood forms above 2800 m.

Patterns of Litter Mass

Litter beneath both species was of the mor type; the boundary with the underlying mineral soil was well defined in most cases, thus separation was relatively easy. Litter accumulated in 01 horizons. As in other coniferous forests of California (cf. Dickson and Crocker, 1953; Zinke and Crocker, 1962), the 02 layer was generally missing; 02 horizons were only found in a few samples beneath hemlock, where the litter/soil boundary was indistinct. The litter surface below hemlock was densely matted, while pine litter was less compacted, as shown by their average shearing strengths (1.75 ± 0.87 kg/cm^2 for hemlock, 0.47 ± 0.33 kg/cm^2 for pine). Litter compaction decreased linearly with elevation for both species ($r = -0.538$ for pine, -0.576 for hemlock).

The litter layer beneath trees was nearly continuous, but its depth, predictably, decreased sharply away from the stems (cf. Zinke and Crocker, 1962; Tiedemann and Klemmedson, 1977). Trends below both species were similar, but the geometric mean litter depth was greater below pine at 0 and 1 m distance from stems, i.e., beneath tree crowns (cf. Mergen and Malcolm, 1955) (Fig. 7). Maximum litter thickness near stems was 24 cm for pine and 15 cm for hemlock; maximum depth at 1 m from stems was 17 cm and 12 cm, respectively. Approximately 30% of the pine sites had no litter at 2 m from stems; about 60% lacked litter at 3 m from stems. The corresponding percentages for hemlock were 45 and 55% at 2 and 3 m respectively.

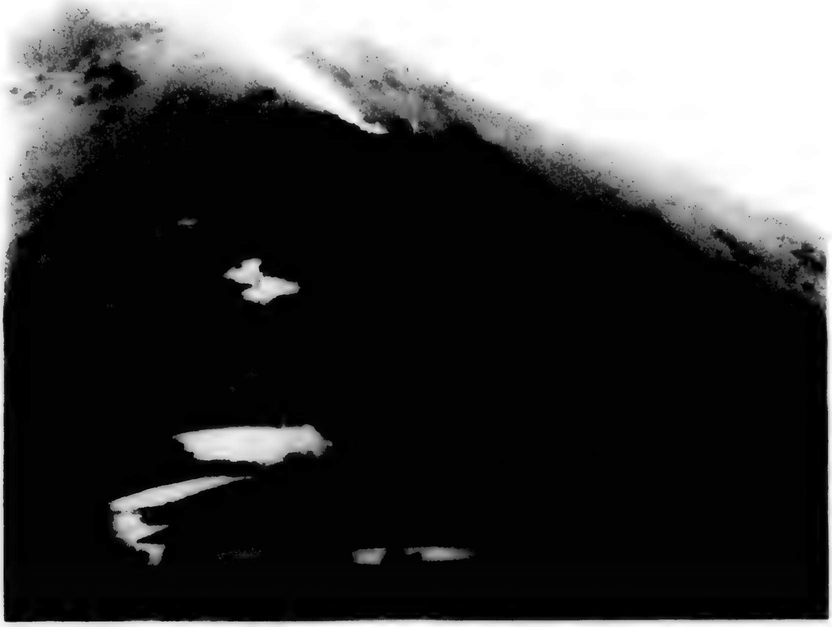


FIGURE 4. Timberline zone on the SE flank of Lassen. Several mountain hemlock islands appear in the foreground; whitebark pine in the middle ground. The summit of Lassen Peak appears in the background. Photograph taken August 1982

Litter depth and weight are obviously related, but different levels of compaction between sites produced a lack of strict correspondence between these two variables (cf. Kittredge, 1940, 1955); weight/depth correlations gave r values of 0.832 for pine and 0.745 for hemlock. Both litter depth and weight (kg/m^2) decreased with altitude in a logarithmic fashion; weight showed a closer association with altitude (r = −0.651 for pine, −0.603 for hemlock) (Fig. 8). A comparison between regression slopes indicates that hemlock sites had lower litter accumulations than pines at the same elevation; this is most evident above 2700 m. The reduction of litter with higher elevation reflects the trend, earlier discussed, of decreasing tree height along the altitudinal gradient.

Even though litter weight was clearly associated with altitude, there was a broad scattering of values (see Fig. 8), not unlike that observed for litter production along latitudinal gradients (cf. Bray and Gorham, 1964, Fig. 1; Jordan and Murphy, 1978, Fig. 1). This suggests that site factors other than altitude affect litter accumulation. Several authors (e.g., Ebermayer, 1876; Bray and Gorham, 1964; Millar, 1974) have commented on the role of slope orientation, through its influence on insolation and soil moisture

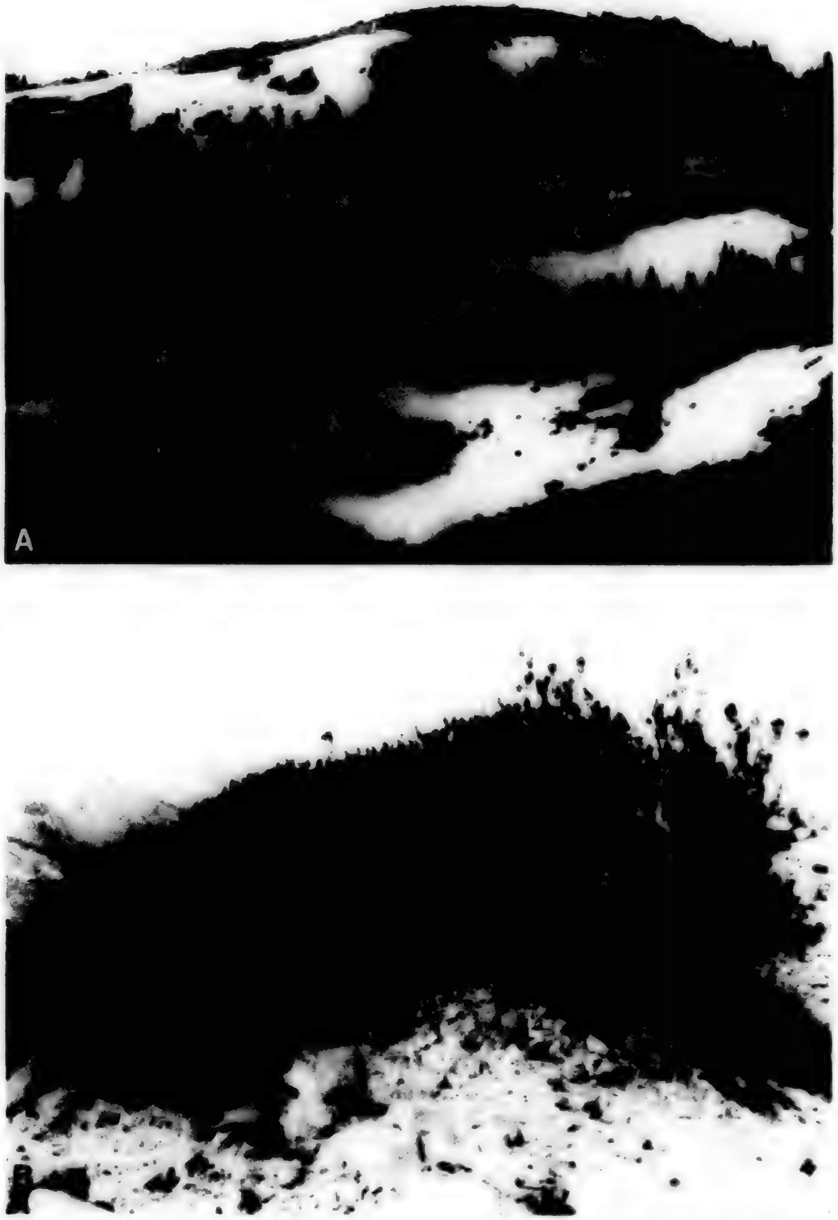


FIGURE 5. (A) Ribbon islands of mountain hemlock 1 km east of Lake Helen; elevation of the lower ribbon is 2625 m. Photograph taken July 1982. (B) Krummholz island of whitebark pine at 2780 m elevation on the SW flank of Lassen Peak. Photograph taken July 1985.

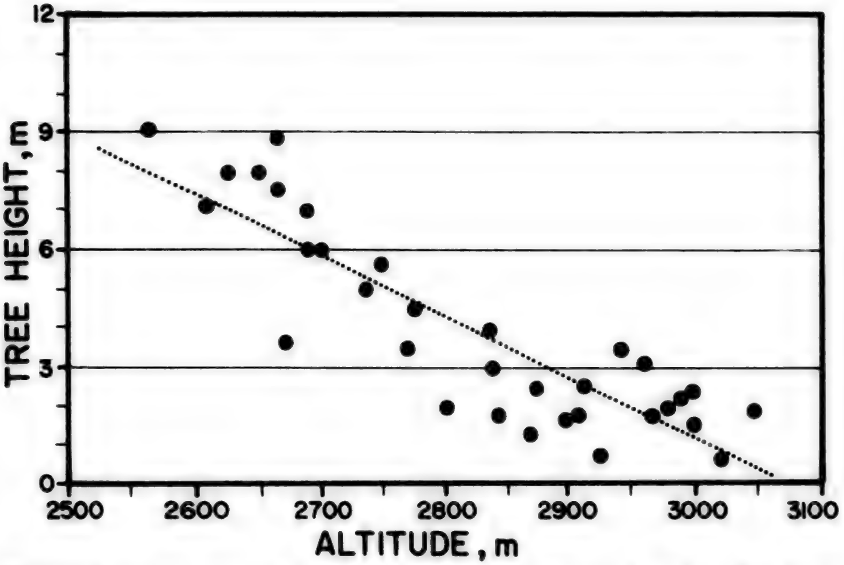


FIGURE 6. Height variation of whitebark pines along the altitudinal gradient. Tree height and altitude in meters. $Y = 48.668 - 0.01578X$, $r = -0.868$, $p = 0.001$, $S_{y.x} = 1.295$.

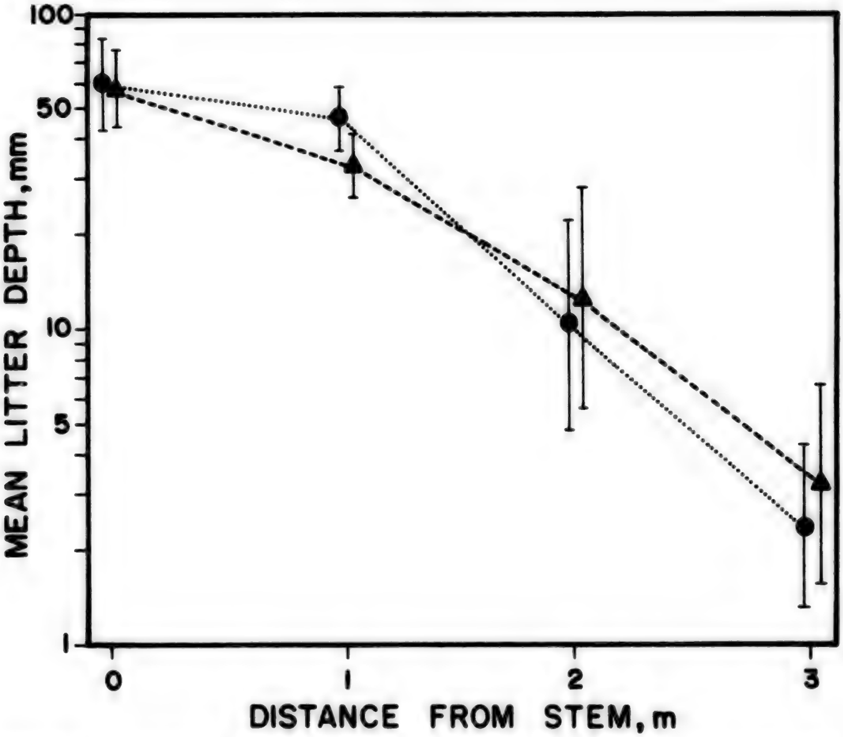


FIGURE 7. Geometric mean litter depth next to tree stems. Circles: whitebark pine, triangles: mountain hemlock. Confidence intervals ($p = 0.01$) shown.

availability, in confounding litter/altitude relationships.

Although most sampling sites at Lassen had a southern exposure, better correspondence between litter weight and altitude was obtained when samples were stratified into two groups (W to SW, and E to SE) (cf. Ebermayer, 1876). The highest correlations were found for 15 pine sites on the E-SE flank ($r = -0.740$ at the $p = 0.002$ significance level) and for 18 hemlock plots on the W-SW slope ($r = -0.701$). Therefore, when the effect of aspect is removed from the data, altitude can "explain" up to 55% (r^2) of the observed variability of litter weight below pines and 50% beneath hemlock. Comparison of samples within altitudinal belts revealed that sites on the SW flank had generally higher litter accumulations than SE-facing plots, but since the number of samples within any elevation belt and exposure was low (Fig. 3), differences were not statistically significant. Clearly, a denser sampling network would be needed to substantiate these results.

Organic Matter Content

The organic matter content of soil samples taken beneath both tree species was related to the amount of litter accumulation ($r = 0.666$ for pine, 0.738 for hemlock). When the effect of aspect was removed, correlation coefficients increased to maximum values of 0.744 for the 15 pines on the E-SE flank and to 0.792 for the 18 hemlocks on the W-SW slope. Thus, litter accumulation accounts for up to 55-63% of the observed variation in soil organic matter. Because of this association, organic matter also drops logarithmically with elevation ($r = -0.677$ for pine, -0.633 for hemlock) (Fig. 9). When samples were stratified by aspect, r rises to -0.712 ($p = 0.002$ level) for the E-SE pines and to -0.770 for the W-SW hemlocks. In all cases, the regression slope for soils below hemlock is steeper than that for soils under pine, showing that the higher litter accumulations below the latter result in correspondingly higher organic matter concentrations (Fig. 9).

Organic matter content of the 35 alpine samples decreases similarly with elevation, but their r value (-0.814) is higher than those for samples under trees. Regressions for separate mountain flanks gave similar values for the W-SW slope, but an $r = -0.868$ for 16 soil samples on the E-SE aspect; this explains up to 75% of the observed variability of organic matter in alpine soils. Fig. 9 shows that alpine samples had considerably lower organic matter percentages than samples beneath trees at the same altitude (cf. Perez, 1987). The regression slope of the alpine soils closely parallels that of pine samples. The reason for this is not apparent; it could simply be coincidental or, alternatively, may point to a dependency of the alpine soil's organic matter on the supply of pine litter, easily blown away from nearby tree islands.

The Effect of Organic Matter on Bulk Density

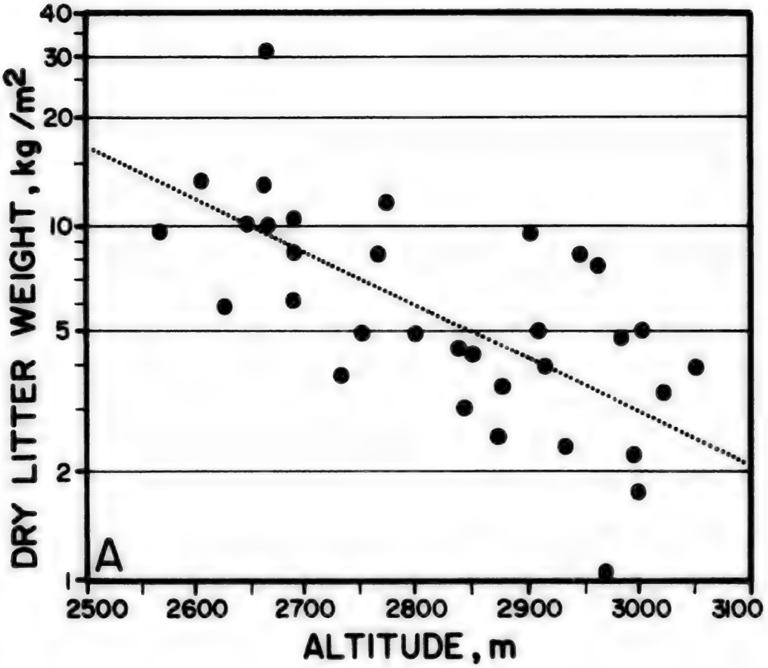
Although a detailed analysis of soil properties along the Lassen gradient awaits completion, bulk density was selected here to demonstrate the important effect of litter on soil properties. Regressions between organic matter and bulk density for soil samples in pine, hemlock, and alpine sites gave r values of –0.950, –0.938, and –0.724, respectively. Clearly, the effect of organic matter was higher in tree islands than in alpine soils, due to low levels of organic matter in the latter—usually less than 6%. A regression of all the samples gave an $r = -0.913$, which accounts for 83.4% of the bulk density variation (Fig. 10). Alpine soils had an average bulk density (± S.D.) of 1.165 ± 0.202 g/cm^3; only one value was below 0.85 g/cm^3. Hemlock and pine soils had mean bulk densities of 0.814 ± 0.324 and 0.910 ± 0.291 g/cm^3, respectively; both reached densities as low as 0.35 g/cm^3. Because of its dependence on organic matter, bulk density also increased with altitude within the three sampling series, with r values similar to those reported for litter and organic matter.

DISCUSSION

The amounts of litter accumulated below trees at Lassen clearly depend on the balance of litter production and litter breakdown, but since neither was measured, it is not known which one contributed more to the observed trends in litter mass. Because temperatures decrease along the altitudinal gradient, plant productivity and litterfall should decline with elevation. The higher temperatures on the SW flank should also produce a somewhat greater productivity. Soil moisture also declines upwards, thus almost certainly causing lower productivity rates in the drier sites at higher elevations (cf. Walsh and Voigt, 1977). The different stature of tree growth-forms at Lassen clearly results primarily from differences in adjustment to climate (Clausen, 1965), not in age. However, the larger trees, with their greater crown biomass, produce higher amounts of litter than trees with smaller crowns. Most trees sampled were in all likelihood old enough to have reached a stable primary productivity and exhibit no further increases in litter accumulation below them. However, some trees at the higher elevations were very small, and the thin litter layers below them may simply reflect their youth.

Decomposition rates decrease at lower temperatures following a linear function (Shanks and Olson, 1961), thus slower litter breakdown would result at higher altitudes. Moisture content below saturation levels is positively correlated with decay rates, therefore decomposition should occur faster in the moister, lower timberline sites (Walsh and Voigt, 1977) and on the E-SE flank, where moisture is higher due to lower evaporation rates and larger water supply from snowdrifts, which persist on this slope until mid-summer.

Pine litter, because of its high lignin content, decomposes slowly (Millar, 1974; Meentemeyer, 1984). Litter below whitebark pines at Lassen was more abundant than under hemlock. Many studies comparing



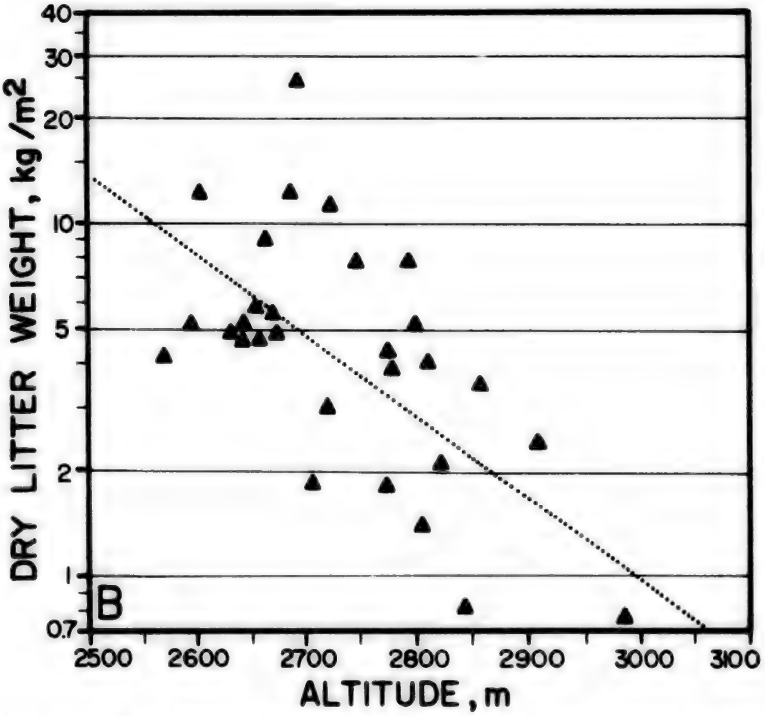


FIGURE 8. Dry litter weight (kg/m²) below whitebark pine (A) and mountain hemlock (B) trees along the altitudinal gradient. A: $\text{Log}_{10}Y = 7.819 - 0.00144X$, $r = -0.651$, $p = 0.001$, $S_{y.x} = 0.237$. B: $\text{Log}_{10}Y = 9.614 - 0.00218X$, $r = -0.603$, $p = 0.001$, $S_{y.x} = 0.286$.

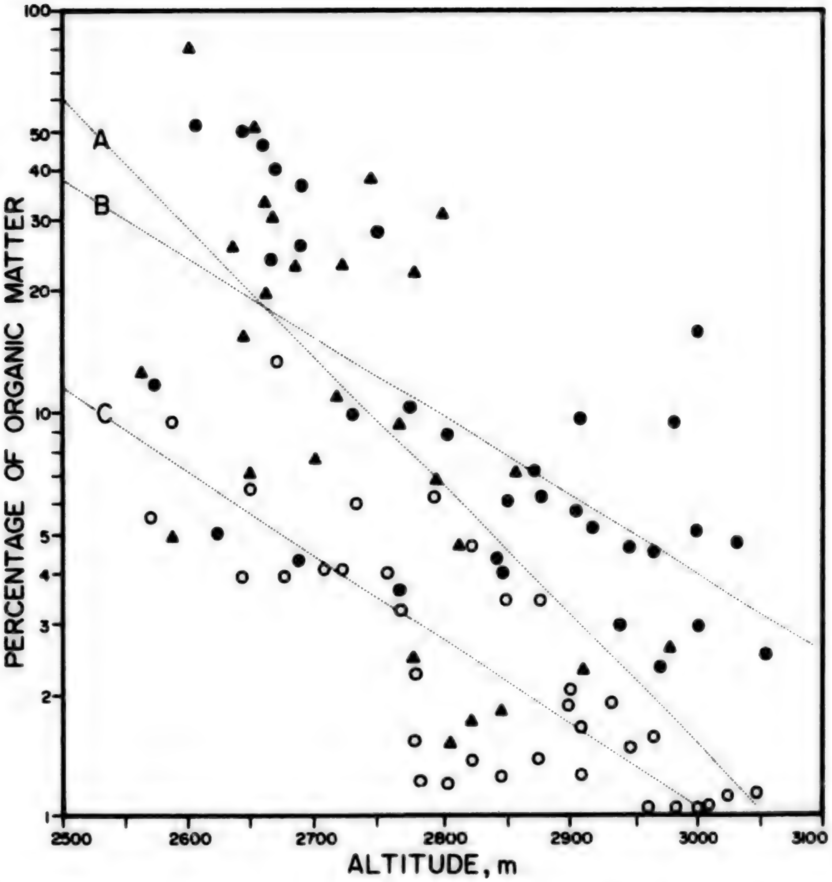


FIGURE 9. Percentage of organic matter of soil samples along the altitudinal gradient. Dotted lines indicate the regression slopes: A and dark triangles = mountain hemlock, $\mathrm{Log}_{10}Y = 9.831 - 0.00322X$, $r = -0.633$, $p = 0.001$, $S_{y.x} = 0.390$. B and dark circles = whitebark pine, $\mathrm{Log}_{10}Y = 6.499 - 0.00197X$, $r = -0.677$, $p = 0.001$, $S_{y.x} = 0.304$. C and light circles = alpine sites, $\mathrm{Log}_{10}Y = 6.033 - 0.00200X$, $p = 0.001$, $S_{y.x} = 0.185$.

pines with other conifers (Kittredge, 1940, 1955; Dickson and Crocker, 1953; Mergen and Malcolm, 1955) have found similar results; some authors have ascribed differences in litter amounts found below different conifer species to differences in rates of breakdown (Mergen and Malcolm, 1955; Shanks and Olson, 1961). The pronounced increase in litter thickness next to the tree stems could also be explained by differences in decomposition rates, since bark litter, which decomposes more slowly than needles, builds up a thicker organic layer (Zinke and Crocker, 1962). Alternatively, these differences in litter depth may partially result from greater wind disturbance outside of tree crowns (cf. Zinke, 1962;

Tiedemann and Klemmedson, 1977).

The high correlation values of the regressions of bulk density on organic matter evidence the significance of litter-derived organic matter in soil development at Lassen. Figure 10 shows, however, how bulk density is markedly reduced with modest amounts of organic matter, but further increases of organic debris only result in minor bulk density changes (cf. Dickson and Crocker, 1954). This relationship suggests that alpine soils are greatly altered by small additions of litter blown from islands; only a 5% organic matter content is needed to reduce bulk density from 1.5 to about 0.95-1.0 g/cm^3 (see Fig. 10). Other soil properties dependent on organic matter may not necessarily follow the same exponential relationship. Since variations in bulk density strongly influence the porosity, permeability, percolation rates, and available water capacity of soil, the differences observed at Lassen clearly ought to have important implications for soil development there. From the data obtained, it would appear that hemlock, despite its lower litter accumulation and resultant soil organic matter, is able to effect a greater change on soil bulk density than pine (cf. Mergen and Malcolm, 1955).

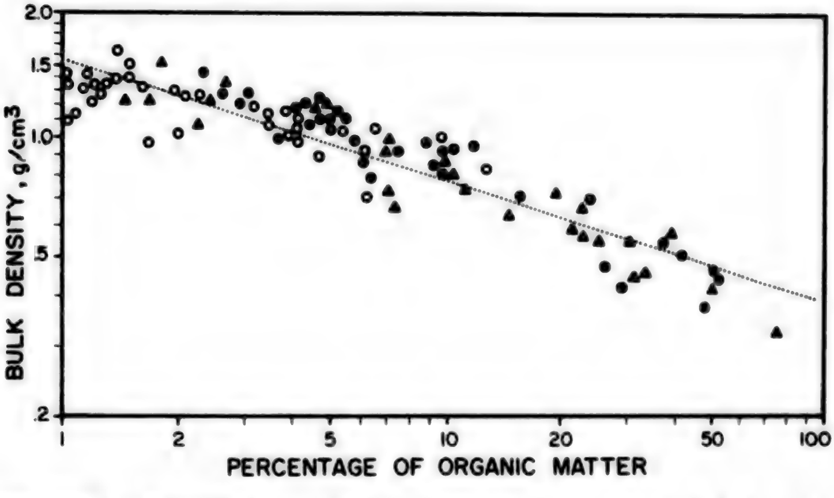


FIGURE 10. Relationship between the percentage of organic matter and the bulk density of soil samples. Key for symbols as in Fig. 9. 100Log10Y = 2.1887 − 0.29746Log10X, r = −0.913, p = 0.001, $S_{y,x}$ = 0.066.

CONCLUSIONS

The main findings of this study can be summarized as follows:

1. Tree height decreased with altitude for both whitebark pine and mountain hemlock. The reduction of tree stature was associated with a

change in growth-form, which results from differences in adjustment to climate.

2. Litter thickness decreased sharply away from tree stems. Litter accumulation outside tree canopies was very thin or nonexistent.

3. Litter thickness and (dry) weight per unit area dropped with elevation. Litter accumulation below pines was usually greater than under hemlock.

4. The organic matter content of soils was dependent on the litter accumulated above; it thus decreased with elevation. Organic matter percentages were higher below pines. Alpine soils away from trees had small organic matter contents.

5. Organic matter markedly influenced the bulk density of soils. This relation was exponential, with small amounts of organic matter resulting in substantial changes; thus, even alpine soils were significantly affected by organic matter. Hemlock litter appeared to produce a greater change on bulk density than pine litter.

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Collecting and Using Soil Survey Information in Park Management: Examples from Redwood National Park

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Abstract. The soils of Redwood National Park are being mapped at two scales for use in site-specific watershed rehabilitation. At a given location, soil drainage is a valuable indicator of landslide susceptibility. Soil texture, depth and rock content affect revegetation success and susceptibility to surface erosion.

Larger map units, based on data from the detailed maps, are used for general park planning. Erosion control and reforestation are major interpretive elements in the general survey. For each map unit there are recommendations on installation, maintenance and environmental protection associated with roads, trails, campsites, structures and human waste disposal. The park's soil survey follows the National Cooperative Soil Survey format in order to be consistent with neighboring surveys conducted by other agencies. Through the National Cooperative Soil Survey, the park can help refine regional land use guidelines based on its watershed rehabilitation experience.

INTRODUCTION

A soil survey is primarily a tool for predicting land management needs. Differences in characteristics among soils translate into differences in management. A soil survey contains four basic elements: (1) maps delineating different land units; (2) map unit descriptions; (3) soil descriptions; and (4) management guidelines. Information about the locations of different soils and their characteristics helps both planning and ongoing land management. During the course of a soil survey, land managers, soil scientists and other natural resource scientists benefit by working together and sharing insights on the physical and biological processes common to their different disciplines. Soils record histories of many significant events, including past human occupation, types of vegetation and geomorphic activity. This paper describes a soil survey in progress in Redwood National Park, and discusses soil survey standards and needs of national parks in general.

THE STUDY AREA

Redwood National Park lies along the western edge of the Coast Range mountains in the northwestern corner of California. The soil survey area covers the lower Redwood Creek basin at the south end of the park. The vegetation is mostly old-growth or second-growth redwood forest, dominated by coast redwood (*Sequoia sempervirens*), Douglas-fir (*Pseudotsuga menziesii*) and tanoak (*Lithocarpus densiflorus*). The climate is characterized by year-around mild temperatures with rainy winters and nearly rainless but foggy summers. Mean annual temperature is about 11° C and mean annual rainfall is about 2000 mm. Elevations range from near sea level to 942 m. The mountains are underlain by sedimentary and schist bedrock (Harden et al., 1982) and the terrain is complex, with great variability in relief and soils. Overall, sediment yields from streams in northern coastal California are among the highest in the world, probably due to a combination of frequent, heavy rainfall, unstable bedrock and rapid geologic uplift. Major storms following intensive land use have increased sediment yields in recent decades (Janda et al., 1975).

SURVEY OBJECTIVES

In many national parks, a low-intensity "reconnaissance" soil survey is adequate for planning, provided it is supplemented by more detailed investigations at sites intended for development. However, Redwood National Park has unusual needs because of its land-use history and the scope of its resource management program. The 1978 park expansion added approximately 13,500 ha of steep, mountainous lands disturbed by road building and logging. The park has a large, ongoing program of site-specific watershed rehabilitation to mitigate man-induced erosion (Bundros, 1986) and to speed recovery of native plant communities (Hektner et al., 1981). In addition to logged redwood forests, part of the area consists of prairie and oak woodlands needing management much different than the forests (Reed and Sugihara, 1987). The challenge in designing Redwood's soil survey was twofold. First, each management concern had to be addressed at an appropriate level of site specificity. Second, the separate elements of survey activity needed to be combined into one general purpose, parkwide land management document.

SURVEY PROGRESS

Detailed Soil Mapping

The most intense soil mapping in Redwood National Park supports rehabilitation in the Redwood Creek watershed. Detailed information is needed because soil properties affect rates of erosion and the rate and quality of revegetation. Since complex linkages exist among the geomorphic and hydrologic processes at different points on hillslopes, treatment of any one site may affect many others. Detailed hydrologic and geomorphic maps are thus indispensable tools in planning erosion control. Since the soils are formed via many of the same processes that shaped the landforms, soil profiles show signs of these processes. Thus, soil maps supplement and reinforce geomorphic mapping and interpretation.

In 1982, to facilitate detailed mapping, a "soil shorthand" was developed to differentiate soils in the field. Soils were grouped by the type and arrangement of horizons, color, particle-size class, type of bedrock contact, depth to water-related mottles and depth to bedrock. Categories in the local soil shorthand determine classification at the series level (Soil Survey Staff, 1975, 1987; USDA, 1988a). Use of standard categories and notations maintain consistency among different soil mappers and is a working reminder of the minimum information needed to distinguish and classify individual soils. The shorthand notations provided immediate, temporary categories for classification until new series could be established.

Soil patterns were recorded in the field on 1:6000-scale aerial photographs. Soil transects were recorded along roads and tractor trails in units intended for rehabilitation. Mappers kept a map legend and recorded the presence of a specific soil with a colored line or point. This method is fast and easy to use. Delineations can be as small as 1 mm across the mapping scale, corresponding to 6 m on the ground. Very small areas of problem soils can be identified at this scale for special treatment.

Although straightforward and objective, the shorthand system provided no soil names. Names are important to both mappers and users as a reminder of key properties, management needs and differences among the soils. Names are most useful when not too numerous. Examinations of the soil transects quickly showed which soils were important simply by measuring and adding up like units. When this was done, some categories proved extremely common and many others were relatively rare. By 1983, the more common soils were assigned tentative names and formal series descriptions were written. In addition to their proportions, the transects showed how the different soils were spatially positioned on the landscape, relative to one another and relative to other independently mapped geomorphic and hydrologic features. Popenoe (1987) described the soil series and supporting laboratory data. The following examples of typical forest soil series illustrate their significance in watershed rehabilitation.

Coppercreek Series (Fine-loamy, mixed, isomesic Typic Haplohumults). These soils are the most common in the lower Redwood Creek basin. They are formed in deep colluvium on steep slopes. Their uniformly brown colors indicate good drainage and relatively little hazard of mass failure within the soil profile. Their high organic matter content, loamy textures and moderate rock content make them the most productive forest soils in the area. Both redwood and Douglas-fir seedlings thrive on Coppercreek soils in areas physically disrupted by logging or rehabilitation. The loamy textures and moderate rock content of the Coppercreek soils, however, makes them particularly vulnerable to surface erosion on steep slopes. Once forest litter and root layers are broken, there is little resistance to detachment by rainsplash or running water. Coppercreek soils are particularly susceptible to gullying by diverted streams (Weaver et al., in press). Therefore, in planning rehabilitation treatments, park hydrologists should carefully map all actual and possible paths of surface water and prescribe treatments which reduce water concentrations and minimize their diversion potential.

Lackscreek Series (Loamy-skeletal, mixed, isomesic Typic Haplohumults). These brown soils occur on divides and convex areas in close association with the Coppercreek soils. They are less than 1 m deep to bedrock and much of their volume consists of rock fragments. Areas with convex relief and thin, rocky soils yield little sediment to erosion. On the other hand, thin soils lose a greater fraction of their ability to support plant life for each millimeter that is lost. If thin soils become eroded, restoration of natural soil depth greatly improves vegetation reestablishment. During timber harvest, divides are commonly used for the skidding, loading and temporary storage of logs. The soils are often completely removed or very seriously damaged. During watershed rehabilitation, soil returned to these locations greatly reduces the likelihood of persistent, unvegetated scars on the landscape. With their high rock content and moderate depths, the Lackscreek soils have a limited moisture storage capacity. The typically convex relief and windy exposures increase drought stress. In planting seedlings, the park adjusts the proportion of trees toward Douglas-fir on these sites, and toward use of larger, two-year-old stock, which are better able to withstand drought. The high rock content slows planting. It takes more care to avoid damaging roots. Not every location is suitable, and tree planters may make a number of false digs before finding a good spot.

Devilscreek Series (Fine-loamy, mixed, isomesic Typic Humitropepts). These soils are formed in deep colluvium on steep slopes near creeks and in wet hollows below areas of Coppercreek soils. The color, texture and behavior of the upper layers are much like those in a Coppercreek soil, but the deep, subsoil layers are gray with brown mottles from periodic saturation with groundwater. Devilscreek soils are on slopes that are naturally unstable and landslides are common. Cut and fill failures are much more frequent along roads through Devilscreek soils than through Coppercreek or Lackscreek soils (LaHusen, 1984). During rehabilitation, roadfill is removed more completely wherever there are Devilscreek soils near streams. The slopes are re-contoured to stabilize the gradients and restore routes for surface and groundwater drainage. This work tends to be expensive, but it is also very effective in terms of keeping large amounts of sediment from entering streams. Devilscreek soils are replanted with redwoods. Very few Douglas-fir occur naturally on Devilscreek soils, and since Douglas-fir can be killed by submersion of their roots, planting Douglas-fir is not recommended.

Trailhead Series (Clayey, oxidic, isomesic Typic Palehumults). These are red, well drained, silty clay soils that occur on gentle, upper slopes. They are very old, deeply oxidized, and have relatively low susceptibility to either mass or surface erosion. In most cases, very little work is necessary to control erosion. Both redwood and Douglas-fir seedlings do well.

General Soil Survey

The general soil survey in Redwood National Park is intended primarily for planning of developments, from the present to well past the time that watershed rehabilitation has been completed. Development planning is initially less site-specific, but it requires broader management guidelines. For this purpose, soil maps of the south park area as a whole are being prepared at a scale of 1:12000 and will be reduced to 1:24000. Steepness of slope, vegetation and geologic substratum all affect soil behavior and management, so all these factors were considered during the design of the mapping units. Units were given standard slope gradient classes of 0-15, 15-30, 30-50 and 50-75%. Differences in soil moisture were inferred from the vegetation based on the understory indicators of Muldavin et al. (1981). A two-year study of soil moisture and temperature factors calibrated the vegetation indicators to actual soil moisture patterns (Sturhan, in preparation). Information on bedrock geology was obtained from mapping by Harden et al. (1982) and from field observations. Soil patterns were documented in relation to topography (Popenoe, 1985), and to hydrologic and geomorphic processes (Marron and Popenoe, 1986). Although a number of soils are present, each delineated area is as homogeneous as possible at the mapping scale, in terms of its endogenous controlling factors and behavior and its response to management.

The 1:24000 map units are named by their soil and phase characteristics, following USDA (1983) conventions. Along a typical transect, soil series change within distances that range from a few meters to about 100

m. At 1:24000, it is not practicable to depict individual soils on a map. Instead, groups of soils in close proximity and on similar topography are mapped together. Most of the 1:24000-scale map units in Redwood National Park are "complexes," defined as map units in which the soils have a regular, predictable pattern, but no one series covers 75% or more of the area. Complexes are named after those series that cover 15% or more of the area. The "Devilscreek-Elfcreek complex, 30 to 50% slopes" is an example of a complex in which soils of the Devilscreek and Elfcreek series predominate.

Soil transects, recorded at 1:6000 for watershed rehabilitation, were used to define the internal composition of the 1:24000 map units. Transecting is an efficient survey method (Steers and Hajek, 1979). The great number of detailed samples gave a much better picture of the patterns and variability within and among map units than is usually possible in a soil survey. This detail is conveyed to the user in the text of the map unit descriptions, and examples of typical 1:6000 transects visually reinforce the text. Soil surveys can be used in many ways for planning and management. At Redwood, interpretive guidelines for permanent roads, trails, campsites, structures and human waste disposal are being prepared for each map unit. Two contrasting examples are discussed below.

Trailhead Clay Loam, 15 to 30% slopes. This map unit is 80% Trailhead clay loam. It consists of just a few large, broadly elongated areas on upper slopes. These areas are suitable for park developments. The upper slopes, with their favorable relief and low erosion potential, are preferred routes for roads and trails. Roads and trails should be surfaced because the fine-textured soils have very little bearing strength when wet. During the rainy season, vehicles driven on the bare soil cause deep ruts. Dust seriously impairs visibility when the soils are dry. Graded gravel roads control both problems, are easily maintained, and permit all-weather traffic. Flat benchy areas, common on these slopes, are suitable for dry campsites or other facilities. Depth and drainage are satisfactory for standard septic leach fields. However, there may be slow percolation due to the fine textures. If so, the size of the leach field may be increased for satisfactory performance.

Devilscreek-Elfcreek Complex, 30 to 50% slopes. This map unit is 45% Devilscreek gravelly loam, 20% Elfcreek gravelly loam and 35% inclusions of other soils. It occurs in wet hollows on steep slopes and near streams. Map delineations are often long and linear, with irregular boundaries probably defined by discrete erosional events. The soils are wet, unstable and usually underlain by highly sheared schist. In this environment, roads and trails are very costly to build and to maintain. Environmental safeguards are extremely important due to the proximity to streams. Engineering required for drainage and structures is very expensive. Even with the best engineering, roads and trails may be closed by natural landslides. The least expensive, least damaging routes in the mountains will minimize the distance through such unstable terrain.

DISCUSSION

Park soils are an important natural resource, one which the National Park Service (NPS) is only beginning to investigate. Historically, the federal agencies most active in soil surveys have been those with more intensive (consumptive) land uses, most notably the USDA Soil Conservation Service (SCS) and Forest Service. Among federally administered lands in California, the national parks are conspicuous for their lack of standard soil surveys (USDA, 1988b). As park use increases, soil information becomes more critical. How can the parks catch up?

Soil investigations are time-consuming and expensive. Experience has demonstrated that it is most efficient to cooperate with other agencies and universities. Most agencies share NPS concerns, particularly those relating to recreation and development. Soil surveys give scientists and managers general purpose, map-unit specific information for decision-making. Except for local or regional differences, land-use guidelines can be computer-generated by the SCS from standard soil descriptions and map unit information (USDA, 1983). Surveys following National Cooperative Soil Survey (NCSS) standards can arrange for SCS assistance. The Soil Conservation Service is the lead agency for soil surveys in the United States, and is responsible for the review and correlation of all standard NCSS soil surveys, establishment of soil series, map units and associated information. The National Park Service signed a Memorandum of Understanding with the SCS and joined the NCSS in 1983. Now Redwood's soil survey and all future park surveys can be correlated under the national system. The Soil Conservation Service has already helped Redwood with two field reviews, correlation of nine tentative new series, and special laboratory investigations of nine soils.

Cooperation means the Park Service can help other agencies, too. Soil scientists are concerned with the preservation of soil resources for future generations. Parks are perceived as relatively "safe" locations for baseline natural resources research and monitoring. Redwood had already begun its soil survey when the SCS began mapping the north coast this year. The California Department of Forestry anticipates starting work in the region in 1992. Information already collected at Redwood is being shared with these agencies to improve the overall understanding of soils. By cooperating with the NCSS, the park's mapping efforts will tie smoothly with those conducted by other agencies. In addition, the park can contribute improved regional soil use guidelines based on its watershed rehabilitation experience.

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Monitoring Prescribed Fire Effects on Chamise Chaparral

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Abstract. Chaparral vegetation monitoring guidelines are proposed for use in Sequoia and Kings Canyon National Parks to provide managers with the data necessary to evaluate their ongoing prescribed burning program. The guidelines are based on experience gained in a long-term monitoring program in chamise (*Adenostoma fasciculatum* H. & A.) chaparral. The key steps are: (1) articulating management objectives; (2) reviewing the scientific literature for appropriate sampling techniques; (3) deciding which ecosystem parameters to measure or monitor; (4) determining the level of precision required of the sampling techniques to meet management objectives (i.e., testing for appropriate plot and sample sizes); and (5) providing a framework for evaluating sampling results. Methods for establishing long-term treatment and control plots are also presented. The approach can also be used for other vegetation types or in other areas.

INTRODUCTION

The low elevation foothills of Sequoia and Kings Canyon National Parks present a formidable management challenge. Given the responsibility for preserving natural ecosystem processes, the highly flammable nature of chaparral, and the broadly distributed heavy fuel distributions that have resulted from fire suppression activities, these parks are faced with a management problem requiring the use of prescribed fire.

One of the most important aspects of the fire management program for the foothill zone of Sequoia National Park is the development of an understanding of the effects of variable fire interval and intensity on the dominant chaparral vegetation. Experimental burns and wildfires must be monitored and evaluated to develop such an understanding. This will eventually assure that fire management objectives are met, and that no significant long-term impairments to the ecosystem occur. In addition, before natural fire can once again be the dominant force in the foothill zone, much more information must be gathered on fire behavior and its relation to fire effects.

The guidelines that follow are based on experience gained in a long-term monitoring program in chamise (*Adenostoma fasciculatum* H. & A.) chaparral (Parsons and Stohlgren, 1987; Rundel, 1986; Stohlgren et al., 1989). The general approach suggested here can be easily adapted to other vegetation types or other areas.

The development of chaparral monitoring guidelines requires a clear articulation of management objectives, a review of the pertinent scientific literature for appropriate sampling techniques, decisions on which ecosystem parameters to measure or monitor, determination of the level of accuracy required of the sampling techniques to meet management objectives (i.e., testing the precision of biomass estimating equations and testing for appropriate plot and sample sizes), and the development of a framework for evaluating sampling and monitoring results.

Articulating Management Objectives

Clearly articulating management objectives is not an easy thing to do. The National Park Service's Management Policies (USDI, 1978) provide only a general overall goal: "to manage the natural resources of the National Park System to maintain and perpetuate their inherent integrity." Interpretations of this goal have varied, particularly with respect to fire management activities (e.g., Bonnicksen and Stone, 1982; Parsons et al., 1986).

The Fire Management Plan (USDI, 1986) for Sequoia and Kings Canyon National Parks has more specific objectives, paraphrased as: (1) restoring fire to its natural role in the ecosystem; and (2) reducing heavy fuels where necessary to prevent catastrophic fire and to protect the parks' neighbors, visitors and facilities. Working in tandem with the above objectives are lower-level objectives specific to particular prescribed burns documented in individual "Burn Plans." These might include burning under a narrowly defined "prescription" (specific air temperatures, relative humidity, wind speed and direction, etc.), reducing fuels by some given percentage, and limiting smoke production.

Two additional, implicit objectives are to increase our understanding of park ecosystems and to fully document the impacts of management on those ecosystems. These may be the most important, because meeting them assures that management actions are continually reevaluated and modified accordingly (Parsons et al., 1985).

A REVIEW OF THE LITERATURE

A review of the scientific literature pertinent to the vegetation type in question is an important first step in selecting appropriate sampling techniques. Unfortunately, such a search is likely to find a plethora of techniques designed to address different questions, respond to different objectives, and measure different parameters than those one had in mind. For chamise chaparral stands in southern California, for example, Wakimoto (1978) estimated biomass (fuels) using dimensional analysis techniques based on ice cream cone-shaped shrubs. It was clear from similar work in Sequoia National Park that chamise in the southern Sierra Nevada differed in shape, biomass and productivity (Rundel and Parsons, 1979) compared to chamise of the same age in southern California, and that basal stem diameters provided better estimates of biomass (Stohlgren et al., 1982). Thus, local research results were needed to develop appropriate sampling techniques.

Fortunately, the low elevation foothills of Sequoia National Park have been the focus of a number of ecological studies. These include community descriptions (Baker et al., 1982; Rundel and Lambert, unpublished; Rundel, 1986), and studies of fire history (Parsons, 1981), fire ecology (Parsons, 1976; Rundel and Parsons, 1979, 1980; Rundel, 1986); and shrub biomass and demography (Stohlgren et al., 1982, 1984; Rundel et al., 1986; Parsons and Stohlgren, 1987; Stohlgren et al., 1989). Despite this apparent wealth of baseline information, serious limitations remain in the available database. First, much of the previous research has focused on south slope, chamise-dominated chaparral. Portions of the chamise chaparral monitoring guidelines that follow may thus not be applicable to north slope, ridge top, or mixed chaparral. Second, previous emphasis was placed on woody vegetation effects and little or no information exists on potential fire effects on other ecosystem components (i.e., air quality, fauna, soil chemistry, erosion, herbaceous species, etc.): Thus, while the following guidelines focus on evaluating changes in the dominant vegetation over time, it should be recognized that other ecosystem parameters may also need to be evaluated.

SELECTING APPROPRIATE SAMPLING/MONITORING TECHNIQUES

The choice of parameters to measure and monitor depend on the specific objectives of the monitoring effort, the accuracy and precision required of the techniques, and cost and other logistical constraints.

In this example, the specific objectives of the chaparral vegetation monitoring guidelines are to:

1. Provide methodologies to estimate chaparral fuel loading nondestructively and produce an index of burn intensity; and

2. Detail methods for establishing long-term vegetation plots to describe pre-burn woody vegetation and fuels or "biomass," post-fire shrub mortality, and changes in fuel loading, stand density and shrub species composition.

Choice and Accuracy of Parameters

Given the above objectives, the ecosystem parameters of interest are shrub (or stand) biomass, a fire intensity index, and shrub species composition. While it is relatively simple to develop cost effective ways to measure these parameters, park managers must first decide on the level of accuracy required of the techniques to meet management objectives. In Sequoia and Kings Canyon National Parks, managers have decided that biomass and fuel sampling efforts must produce parameter estimates within 20-30% of the population mean. (T. Nichols, pers. comm.).

Estimating Shrub or Stand Biomass

Estimating chaparral fuels (i.e., biomass) to the level of accuracy required by managers often involves developing species-specific biomass estimating equations using dimensional analysis techniques. Typically, an easily measured parameter, such as stem basal area (Stohlgren et al., 1982; Parsons and Stohlgren, 1987), canopy dimensions (Wakimoto, 1978; Stohlgren et al., 1989), or their combinations (Kauffman, 1986; Hughes et al., 1987) is correlated with shrub dry weight and provides a simple, nondestructive way to estimate the biomass of a shrub or stand. Examples of equations used to estimate shrub biomass are shown in Table 1. Note that in addition to being species-specific, the equations may be stand age or even site-specific. For example, because of the different architecture of young chamise resprouts, canopy dimensions are better predictors of biomass than basal stem diameters for such individuals.

Also note that high coefficients of determination (r^2 values) do not ensure accurate predictive capabilities of the equations (Whittaker and Woodwell, 1968). For a given measured "X" parameter, for example, the estimate of relative error (i.e., the expected error on the estimate of shrub dry weight) may be between 17-63% (Table 1). Increased sampling or multivariate dimensional analysis techniques (see Kauffman, 1986 or Hughes et al., 1987) will be needed to increase the accuracy of the equations with high "e" values.

The equations in Table 1 cannot be expected to be as accurate for a stand of greatly different age or site characteristics. The equations, however, can be easily field tested by subsampling a few more stems or shrubs in the target area and comparing the observed against expected results (see Zar 1974, pp. 209-213).

Once accurate biomass estimating equations are available, stand biomass (or fuel loads) can be estimated by establishing a number of randomly located plots in the vegetation type of interest. Data gathered from several previous studies (Stohlgren et al., 1984; Parsons and Stohlgren, 1987; Stohlgren et al., 1989; Sequoia National Park, unpublished data) show that the density of shrubs in stands 24 years old or older (i.e., since the last

TABLE 1. Biomass estimating equations (y in kg) for dominant species in different age chamise chaparral stands in Sequoia National Park, California. The estimate of relative error, or "e" value was calculated following Whittaker and Woodwell (1968).

Stand Age/Species	Equation	Units of X	Number Samples	r^2	e	Source
3 Year Old Stand						
Adenostoma fasciculatum	$y = 2.47\ X^{0.99}$	cylindrical volume in m^3	28 shrubs	0.88	36%	Stohlgren et al. 1989
24 Year Old Stand						
Adenostoma fasciculatum	$y = 2.47\ X^{0.99}$	stem diam. in cm^2	24 stems	0.91	17%	Parsons and Stohlogren 1987
Ceanothus cuneatus	$y = 0.42\ X^{1.28}$	stem diam. in cm^2	62 shrubs	0.82	57%	Parsons and Stohlgren 1987
Eriodictyon californicum	$y = 0.12\ X^{1.30}$	stem diam. in cm^2	41 stems	0.88	63%	Parsons and Stohlgren 1987
Rhamnus crocea	$y = 0.30\ X^{1.03}$	stem diam. in cm^2	17 stems	0.92	42%	Parsons and Stohlgren 1987
Fraxinus latifolia	$y = 0.31\ X^{1.28}$	stem diam. in cm^2	24 stems	0.91	17%	Unpublished data
>60 Year Old Stand						
Adenostoma fasciculatum	$y = 0.18\ X + 0.06$	stem diam. in cm^2	50 shrubs	0.94	22%	Stohlgren et al. 1982

TABLE 2. Relationship of biomass (Y in kg ha^{-1}) to density (X in shrubs ha^{-1}) of cover (%) for chamise chaparral.

Stand Age in years	Regression Equation	No. Plots	r^2	Accuracy[a]
>60	$Y = 9.73(\text{density})-22{,}400$	10	0.63	± 27%
24	$Y = (1080)\ (\text{density}^{0.37})$	20	0.84	± 18%
15	$Y = (0.06)\ (\text{cover}^{2.54})$	6	0.64	± 33%
3	$Y = (62.07)\ (\text{cover}^{1.13})$	10	0.50	± 42%

[a] Based on estimate of relative error "e" value (Whittaker and Woodwell, 1968) on linear regression.

burn) can be an accurate predictor of stand biomass (Table 2). In younger stands, however, density was a very poor predictor of stand biomass, and although data from line intercept cover transects was a better predictor, it did not meet the accuracy requirement (if 30% accuracy is required) of the managers (Table 2). Thus, in young (< 20 year old) chamise stands, additional sampling or multivariate dimensional analysis techniques (see Kauffman, 1986 or Hughes et al., 1987 for detailed methods) will be necessary to estimate stand biomass. However, since older chamise stands are likely to be scheduled for prescribed fires, only total shrub density data may be needed to estimate stand pre-burn biomass.

Selecting the Appropriate Plot Size

The appropriate plot size for measuring chaparral fuels can be determined by plotting calculations of biomass per unit area against different preliminary plot sizes (Fig. 1), as suggested in Mueller-Dombois and Ellenberg (1974, pp. 48-52). Data were derived from the first two 10 m × 10 m plots established in a 24-year-old stand of chamise chaparral (see Parsons and Stohlgren, 1987) by systematically increasing plot size (i.e., 4 m^2, 16 m^2, 36 m^2, etc.) from one corner of the mapped plot and calculating the running mean biomass and density per unit area. Despite differences in biomass per unit area between plots, the curves tend to level off between 36 and 64 m^2. A plot size of 50 m^2 resulted in biomass per unit

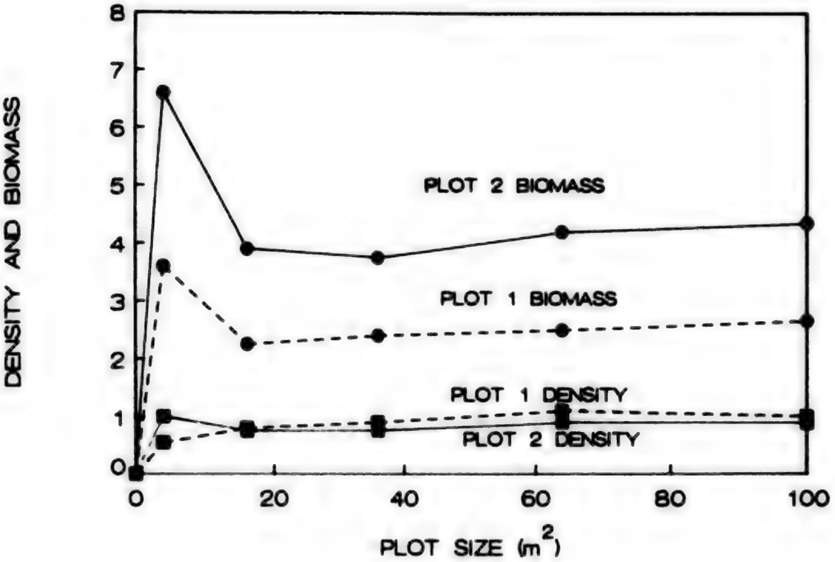

FIGURE 1. Biomass (kg/m^2) and density (No./m^2) of chamise chaparral for various plot sizes (data from two plots in a 24-year old stand; see Parsons and Stohlgren, 1987).

area estimates within 13% and 9% of the biomass in 100 m^2 size plots for plots 1 and 2, respectively, and appears to be adequate for shrub biomass and density measurements. Although smaller plots may be adequate for biomass estimates in younger stands or density estimates in mature stands, long-term monitoring of biomass accumulation requires the larger plot size.

Sample Size Determination

One method of determining appropriate sample size (Cochran, 1977; Scheaffer et al., 1979) is to: (1) determine the percent accuracy (P) desired (i.e., how close to mean biomass does one need to obtain to meet management objectives); (2) calculate the mean (X) and standard deviation (SD) of the biomass of a preliminary sample; and (3) calculate the sample size (n) as:

$n = 4 * SD^2 / d^2$ where $d = X * P/100$.

For data from the first 10 plots established in a 24-year-old stand of chamise chaparral (see Parsons and Stohlgren, 1987), the mean biomass and standard deviation were 308.9 kg/100 m^2 and 83.8 kg/100 m^2, respectively. Given a desired percent accuracy of only 30% of the mean, only 3.3 (or 4) plots would be required to meet management objectives (Fig. 2). However, in monitoring prescribed fire behavior, increased accuracy of

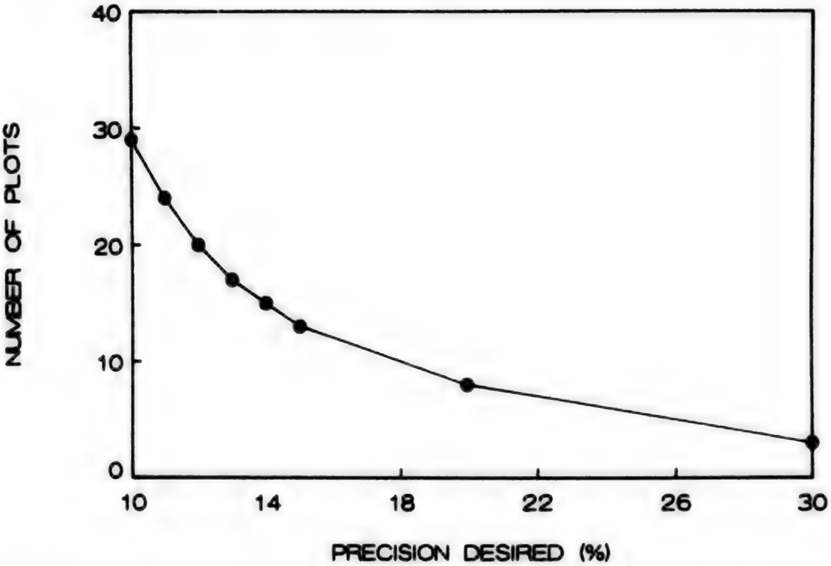

FIGURE 2. Number of plots required under various levels of desired accuracy to estimate stand biomass in mature chamise chaparral (data from ten plots in a 24-year old stand; see Parsons and Stohlgren, 1987).

biomass estimates may be required to estimate flame length or rate of spread. If such is the case, it is easy to recalculate the number of plots required for different levels of accuracy by adjusting the value of "P" in the above equations. As shown in Figure 2, the number of plots required greatly increases as the level of accuracy increases (i.e., as estimates are desired closer to the population mean).

ESTABLISHING LONG-TERM MONITORING PLOTS

The management objectives for establishing long-term, fire effects vegetation plots are to evaluate the fire management program by monitoring changes in fuel loading, stand density and shrub species composition. The important considerations regarding to plot size and the number of plots necessary to evaluate long-term vegetation change have been discussed above. Mueller-Dombois and Ellenberg (1974) suggest, however, that rectangular rather than square or circular plots should be used to reduce variability within plots. For our example of mature stands (> 20 years since last burned) of chamise chaparral, four 50 m^2 (5 m × 10 m) randomly located plots are necessary to produce standard errors of ±30% of the mean for density and biomass. Keep in mind, however, that the number of plots used to describe an area is dependent on the homogeneity of the vegetation (i.e., age species, composition, etc.), physical features of the landscape (i.e., slope, aspect, micro-topographic features, etc.) and the level of accuracy desired for specific objectives.

For chamise-dominated areas, stand age and micro-topographic features (i.e., ravines, rock outcrops, etc.) are probably the most important factors affecting biomass and density homogeneity. Thus, prescribe-burn units should be delineated in single-age stands whenever possible, and atypical ravines, rock outcrops and ecotones should be stratified out of the potential sampling area prior to plot selection.

Pre-burn Sampling

The long-term vegetation plots should be established immediately prior to burning to characterize pre-burn (competition-caused) mortality, species composition and density. Plots should be marked with 2-m-tall rebar stakes with brightly painted tops. The number of shrubs by species (live or dead also noted) should be recorded for each plot. This will provide baseline data on pre-burn biomass (Table 2), species composition and density, all of which can be compared in later years.

Estimating Burn Intensity

A quick and easy approximation of burn intensity (albeit post-facto) can be determined by calipering the diameter of 30-50 randomly selected terminal branch tips of the shrubs in post-burn stands. This provides a burn intensity index to evaluate fire effects within and between stands and a gross approximation of percent fuel reduction for chamise-dominated stands (Table 3).

TABLE 3. Approximation of burn intensity and percent fuel reduction based on mean terminal branch diameter in post-burn stands.

Mean Branch Diameter(mm)	Burn Intensity Index	Percent Fuel Reduction[a]
0-10	Low	< 25
10-25	Moderate	25-75
> 25	High	> 75

[a] Based on proportion of foliage to stem biomass in a 24-year-old stand (Parsons and Stohlgren, 1987).

One-year Post-burn Sampling

Each long-term plot should be revisited one year after the burn to determine fire-caused mortality. This is done by counting the number of resprouting (burls, not seedlings) in each plot and subtracting competition-caused mortality from pre-burn density data. Typical values for thinning-caused mortality in mature stands are 5-15%, while fire-caused mortality is 15-30% for summer and fall fires (Stohlgren, 1985) and much higher for spring fires (Rundel et al., 1987). It is important to continue gathering information on the relationship of burn intensity to mortality to further refine burning prescriptions in future years.

Fuel loading in one-year post-burn stands will likely consist of small shrub resprouts, subshrubs, herbs and grasses (Rundel et al., 1987; Stohlgren et al., 1984). Estimates of fuel loading can be achieved by clipping, drying and weighing several randomly selected 1-m^2 plots in the burn area. The exact number of plots needed should be determined as described above in "Sample Size Determination."

Five-year Post-burn Sampling

After five years, each long-term plot should be revised to count the number of established seedlings and resprouts by species to evaluate species composition change. For sprouting species, the "replacement potential" of fire-killed burls can be quantified as:

$$\underset{(\%)}{\text{replacement potential}} = \frac{\text{\# seedlings}}{\text{\# dead burls}} \times 100$$

For obligate seeding species (e.g., *Ceanothus cuneaus, Arctostaphylos viscida*), the number of pre-burn individuals replaces "# dead burls" in the equation. This type of information can be used to quantify such major shrub species shifts as would characterize type conversion from chamise to ceanothus or grassland. Fuel loading can be estimated similarly to one-year post-burn stands, but plot size may need to be adjusted (see Plot Size Determination) as well was the number of plots (Stohlgren et al., 1989). Less accurate estimates might be obtained by using live intercept cover data and the regression equation in Table 2 for 3-year-old stands. Alternatively, the multivariate dimensional analysis techniques described by Kaufmann (1986) could be used to improve biomass estimation equations, depending again on the priority of such information to managers.

If time, money, and expertise are available, and if the introduction of alien species becomes a higher research priority, analysis of grass and herbaceous species composition should also be conducted at this time.

Fifteen-year Post-burn Sampling

Each plot should be again revisited after 15 years for the final assessment of changes in species composition and density. Here, the number of live shrubs by species are tallied. For each plot, and each species, the percent change in shrub density is calculated as follows:

$$\begin{array}{l}\text{\% change}\\ \text{in shrub} = \\ \text{density}\end{array} \frac{\text{\# post-burn shrubs} - \text{\# pre-burn shrub}}{\text{\# pre-burn shrubs}} \times 100$$

The mean percent change in the shrub density can be calculated for each burn, and thus, the long-term vegetation plots provide park managers with information on the effects of prescribed fire on species composition changes over time. Fuel loading can be estimated by using existing cover to biomass equations (Table 2) or by using the multivariate techniques described by Kauffman (1986) or Hughes et al. (1987).

Establishing "Control" Plots

Control plots must be established if managers are to be able to evaluate the effects of the alternative management action: *not* prescribe-burning certain areas (i.e., for boundary or visitor safety reasons). In chamise chaparral, it may be particularly important to evaluate the continued effects of fire suppression since shrub (burl) mortality and stand senescence are likely to increase with stand age (Rundel and Parsons, 1979) and vegetation type conversions would be more likely following fire.

Three to five long-term control plots should be randomly placed in similar vegetation (i.e., with similar stand ages, species composition and biomass) as in prescribe-burn units. The number of live and dead shrubs for each species should be recorded at five year intervals for 15-20 years to quantify competition-caused morality. Qualitative notes on the percent dead material per live shrub should also be recorded as this may affect fire behavior.

DISCUSSION

The proposed monitoring program concentrates on the measurement of fuel loading and detection of gross changes in shrub species composition and density. As the data set from this monitoring program develops, so will our understanding of the ecosystem. We may learn to differentiate between species shifts (i.e., from chamise to ceanothus or grassland) caused by fire (i.e., caused by early or late season burns or more or less frequent burns in the same area) from those that might have naturally occurred in marginal habitat at ecosystem boundaries. It may take several years and numerous prescribed fires before we can fully evaluate our management actions.

The establishment of "control" (or untreated) plots is essential to evaluate a park's fire management program. Such plots are not "control" plots in the laboratory sense and may not directly serve as paired comparisons to treatment (burn) plots, but without them managers will be unable to evaluate the effects of continued fire suppression in some areas. Furthermore, information on thinning-caused mortality, stand senescence, and fuel accumulation gained by establishing control plots in mature stands could be used to prioritize prescribed burns. Such plots would not necessarily be left unburned indefinitely, and could later be incorporated into prescribe-burn units to evaluate the effects of fire in long-unburned stands.

Despite the existing information base, an experimental approach should be taken when prescribe-burning chaparral. Data from the monitoring effort should be evaluated in a timely manner to revise burning prescriptions as necessary to meet management objectives.

Application of these guidelines to other areas (i.e., north slope and mixed chaparral in Sequoia National Park, other parks, etc.) will only be possible after accurate estimates of fuel loadings (biomass estimating equations; Table 1) are developed for the dominant species (or growth forms or age classes).

In the future, park managers will likely need detailed information on the spatial *and* temporal distribution of prescribed fires to learn more about the integrity of different age stands as fuel breaks. They also need this information to learn more about the spread of natural fire within and between different vegetation types along the elevational gradients in Sequoia and Kings Canyon National Parks. It is therefore essential to develop a detailed geographic-based information system for the foothill areas of Sequoia and Kings Canyon to aid in the spatial and terrain-dependent aspects of prescribed fire planning.

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Fire Management Policy and Boundary Effects on Parks: Lassen Volcanic National Park— A Case Study

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Abstract. This paper represents one of several studies examining the role of boundaries in the protection of national parks (Schonewald-Cox, 1987). In this work we are specifically interested in what roles the administrative boundary and associated jurisdictional authorities have in the management of fire. This analysis compares the fire management agreement between Lassen Volcanic National Park and Lassen National Forest, with their responses to the 1987 Snag Fire within Lassen Volcanic NP. The combined effects of temporary fire suppression in northern California and the threat of the Snag Fire expanding into commercial forest led to a management response to stop the spread of the fire towards the joint Park Service-Forest Service boundary. With the aid of the Susanville Interagency Fire Center, the park responded to the danger of fire spreading onto land under Lassen NF jurisdiction. We attempted to describe the policy-related generated edge reflecting the impacts on fire management by the differing management objectives of the Park Service and Forest Service. This led to our identifying an "action-response" zone associated with the Snag Fire. We estimated that inside this action-response zone, interagency forces are predisposed to enter the park to assist in extinguishing a fire, thus leaving 39% of the park habitat under the sole fire management control of Lassen Volcanic NP. We concluded that the park does not have complete control over fire management in a significant portion of the park. This region behaves as a generated edge reflecting adherence to jurisdictional authorities and fire management policy. We suggest that it is counter-productive for protection of the natural fire-dependent ecological processes within the park. The establishment of a fire management zone acting as a buffer between differing institutional objectives for vegetation and fire is suggested as a possible solution to the intrusive and deleterious qualities of this "edge." Such a zone, which does not already exist, should consist of land owned by both the park and adjacent forest. A shared area of fire management compromise might reduce future contradic-

tions of policy and management, as were observed here, regardless of usual or unusual fire-producing conditions.

INTRODUCTION

Conflicts in policy, or management adherence to policy, between adjacent jurisdictional authorities can be expected to induce ecological change. This change, which can be brought on by small differences in values, may not be confined to the boundary region where the jurisdictions and associated differences meet (Schonewald-Cox and Bayless, 1986). This is how impacts can enter parks and effectively reduce the area the park is able to protect (Schonewald-Cox, 1987; Diamond et al., 1987). All vegetation undergoes continuous changes as it is exposed to various disturbances (Drury and Nisbet, 1973). Fire is one of the most important causes of community change at the secondary successional level where a seed bank is already present. Disturbance as a result of fire is a natural process which liberates nutrients, provides habitat for new species, and creates a mosaic of various stages of succession (Agee, 1987; Hemstrom and Franklin, 1982; Sumner, 1978). Since national parks attempt to maintain natural evolutionary process such as succession, the management of fire has become a key technology for the promotion and control of vegetation changes (Parsons, 1977; Bonnicksen and Stone, 1985), as has been illustrated in Everglades (Robertson, 1953) and Sequoia and Kings Canyon (Kilgore, 1973 and 1975) National Parks. Despite these facts, fire management problems can arise when fires cross or threaten to cross administrative boundaries, particularly when the land owners or managers place different values upon fire. This is why the National Park Service (NPS) and other agencies have written agreements for response to fire (USDI/NPS, 1984). This interagency cooperation is essential for the productive management of ecosystems dependent upon fire.

Natural resource management plans of many parks, including Lassen Volcanic NP, generally specify that fire is to be managed as a natural controlling agent in the forest ecosystem (USDI/NPS, 1981:45). It is the policy of the National Park Service that lightning-caused fires are to "be permitted to continue to influence the ecosystem if truly natural systems are to be perpetuated" (USDI/NPS, 1978:IV-13). We were concerned with what roles the administrative boundary and associated jurisdictional authorities played in the management of the 1987 Snag Fire within Lassen Volcanic NP. By comparing the fire management agreement between Lassen Volcanic NP and Lassen NF with their responses to the Snag Fire, we examined how management decisions were made during this fire and their potential effect on fire-dependant habitats within the park. The Snag Fire covered 830 acres (3.4 km^2) and burned from 30 August to 15 November 1987. This fire was particularly interesting because it involved interagency cooperation during one of the most severe fire seasons in northern California's recent history.

METHODS

We conducted detailed reviews of the Snag Fire report, examining the history of the fire, the NPS Management Policies (1978), and the Lassen Volcanic NP/Caribou Wilderness Fire Management Plan (1984) for comparison of the established fire management policies. Subsequently, we digitized maps of the Snag Fire and spot fires, the park administrative boundary, Snag Lake, and each fire management zone within the Park and Caribou Wilderness (using AUTOCAD software). Boundaries, zones, and landscapes and topographic features were each digitized on separate layers. We then used the maps and software to compare locations and functions of policy and applications of fire response methods within the park, and with respect to location of adjacent forest resources (Fig. 1)

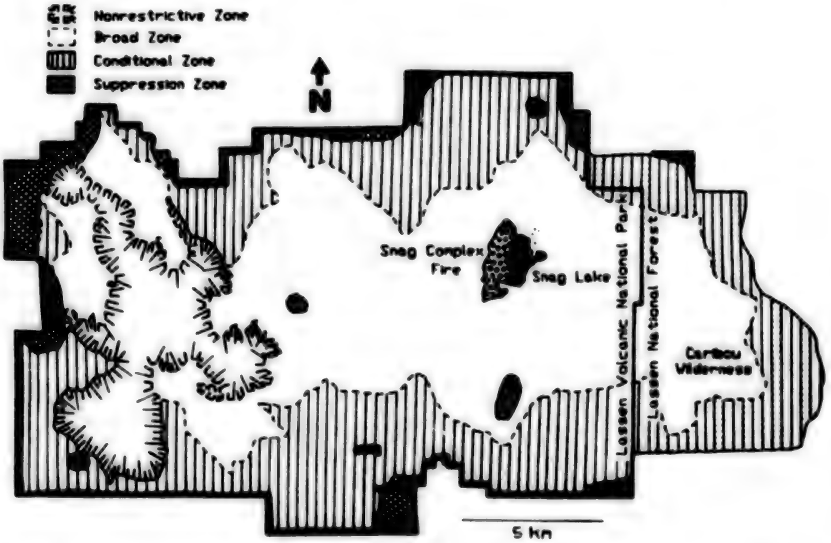

FIGURE 1. Relation of fire management zones to landscape features of Lassen Volcanic NP and Caribou Wilderness Area. Zones are taken from USDI/NPS (1984) and A. Denniston, Lassen Volcanic NP Division of Natural Resources Management (pers. com. 22 March 89). The four black dots northeast of Snag Lake represent spot fires.

OBSERVATIONS

Fire History

Three small fires in the vicinity of Snag Lake, Lassen Volcanic NP, were ignited during a lightning storm which passed over the park on 30 and 31 August 1987. These fires started on an east-facing slope dominated by jeffrey pine (*Pinus jeffreyi*), lodgepole pine (*Pinus contorta*), and red fir (*Abies magnifica*). At the time of the fires, a temporary program of total fire suppression was operating in the park due to the lack of fire control resources and severe smoke conditions in northern California. Attempts to extinguish the fires were unsuccessful and they burned together to form the Snag Fire.

Because the large number of fires limited available fire control resources, no assistance from Susanville Interagency Fire Center (SIFC) could be dispatched to help extinguish the early fire. When the fire grew due to high winds, it ignited four spot fires up to one mile ahead of the main fire. This was close to the park's northwestern boundary. The National Park Service Incident Commander became concerned that the fire might reach national forest land and ordered interagency fire fighting resources (NPS Incident Commander, pers. com., 1989). Consequently, the SIFC (without National Park Service representation) dispatched resources as they became available within the priorities of the fires being fought at that time. With the combined forces, the fire was brought under control and contained within the Snag Lake area. The fire did not threaten human lives or structures, but it did threaten the only known Lassen Volcanic NP nest of a pair of bald eagles. This nest was within the perimeter of the fire, but was protected.

Analysis of Fire Management Zones

For our analysis of fire management zones, we were interested in an area of the park we termed as the action-response zone. We defined this zone as the area inside the park where actions by neighboring land managers on park fires can be expected to influence the flora and fauna within the park. For instance, Lassen NF became concerned for their timber when small spot fires ignited in front of the Snag Fire toward the national park-national forest boundary and consequently, the SIFC dispatched fire fighting resources. This zone was represented by digitizing a hypothetical boundary inside the park boundary at a distance roughly equal to the distance (approximately 2.5 km) from the spot fires to the nearest park administrative boundary (Fig. 2). The approximate areas of the park, each fire management zone, and the region outside of the action-response zone (i.e., the park core) were derived and the percentages of these areas maintained within the park were calculated.

We attempted to measure the potential land that comprised the functional fire suppression zone, including both national park and national forest land (i.e., action-response zone). The suppression of the Snag Fire was stimulated by conditions within the park and the immediate area, and by the SIFC's response when the fire came within a critical distance to the

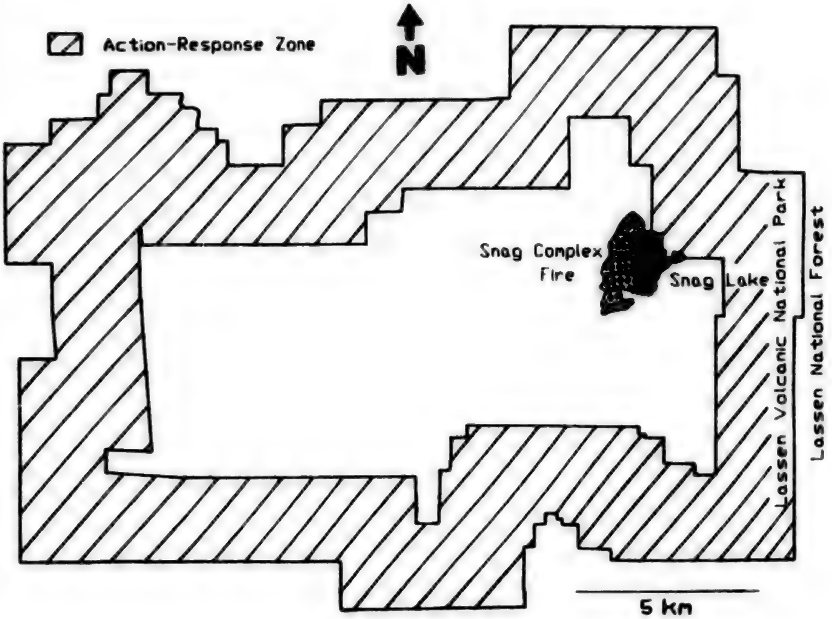

FIGURE 2. Lassen Volcanic NP action-response zone. The width across the zone represents the distance (approximately 2.5 km) from the National Park and National Forest boundary at which the Susanville Interagency Fire Center first sent resources to assist park personnel in suppressing the Snag Fire. The four black dots northeast of Snag Lake represent spot fires.

national forest land (approximately 2.5 km) and burned with a critical intensity (i.e., long-distance spotting under high wind conditions). These two critical factors have a different meaning for both agencies, depending on what the fire threatens and on the particular conditions surrounding a fire.

Even if the action-response zone was not evenly distributed along the entire park boundary, as we have assumed for this particular case, then a functional area of suppression would still occupy some portion of the park's habitat. The remaining area would be within the confines of the park's uncompromised control and protection. In this case, the core of the park is the habitat where the park has significant control over fire management decisions. This area amounts to approximately 39% [41,000 acres (166 km^2) of 106,000 acres (429 km^2)] of the park's habitat.

When we examine the established fire management agreement, we find that only 13% [14,000 acres (57 km^2)] of the park's total acreage is completely subject to the natural influences of fire. This is the Nonrestrictive Fire Management Zone of the Lassen Volcanic NP/Caribou Wilderness

Fire Management Plan (1984). Most of this nonrestricted region is covered with rock, and all natural fires are allowed to burn until they threaten endangered species, people, or structures, or create unacceptable air quality conditions. The fire policy in this zone closely mimics the natural fire regime of the ecosystem, but is spatially limited. The rest of the park's habitat is subject to some prescribed fire suppression, depending on the degree to which conditions are unfavorable for fire.

CONCLUSIONS

One of the primary goals of the Lassen Volcanic NP/Caribou Wilderness Fire Plan (1984) is to "allow fire under prescribed conditions to resume its [fire's] natural role in the Lassen and Caribou Wilderness ecosystems." This goal is being met in some areas, but not in a large portion of the park's habitat. The Nonrestricted Fire Management Zone is the only region where, under most circumstances, natural fires are allowed to burn without the interference of fire suppression forces. Consequently, this is the region of the park's habitat where fire functions normally as an influential natural process. As fire is presently managed in the park, both the park and Lassen NF have agreed to the size and location of this zone as well as the other fire management zones. As a result, both agencies share responsibility in how the park is managed with respect to fire, and both are subject to the consequences of that management.

The actions of the park and Lassen NF during the Snag Fire of 1987 demonstrated a discrepancy between their actions during the actual fire and their fire agreement and objectives. Lassen NF and the park became concerned when the fire reached a certain distance (approximately 2.5 km) from the national park-national forest boundary and thus SIFC, at the park's request, took action to prevent further spread of the fire. To this extent the fire management objectives of Lassen NF conflicted with the policy of perpetuating natural processes favored by the park, creating an intrusive generated edge within Lassen Volcanic NP. If we treat this as a generated edge as defined by Schonewald-Cox and Bayless (1986), then an action-response zone (functionally different from the established Conditional Fire Management Zone), which governs the fate of fire and ecological succession in that area, exists. Consequently, the habitat and organisms within this zone which rely on fire will not experience the natural cycles associated with fire as in other areas which are subject to periodic burning, unless the park begins to utilize prescribed burns within and outside the park. Otherwise, succession without fire could develop major changes in vegetation species abundances and diversity, which is contrary to the original intentions set down in the policies and protection objectives of the park. Furthermore, prescribed burning would help return fuel loads to their normal levels and reduce the potential for extreme fires as suggested by Parsons et al. (1986). Though the prospects for restoring fire to its long-term role seem shaken by the recent events at Yellowstone National Park, the dependency of vegetation mosaic patterns, fuel loadings, and burn patterns on fire will remain.

Further analysis of the action-response zone in Lassen Volcanic NP should be directed to another fire, referred to as the Badger Fire. This fire, which occurred in 1984, may have had a significant influence on the management of the Snag Fire and consequently, the creation a functional fire suppression region. It started in the park, but escaped prescription and crossed onto national forest land. The potential impact of the Badger Fire on the management of the Snag Fire was not examined in the scope of this study, but would be an important part in the continued review of the fire management program of Lassen Volcanic NP and the cooperative management of future park fires.

In addition to fire history, park size is another important aspect of boundary analysis which needs to be examined more closely. The effects of conflicting management policies between parks and neighboring land managing agencies on the parks' habitats and the species they contain would be expected to vary with park size. The smaller the park, the more perimeter relative to area is exposed to other agency policies. Thus, the influence on park habitat by other land managers is most critical for the smaller national parks. These parks should be the first to incorporate action-response zones in their management plans. Further studies that compare the presence and effects of action-response regions in small and large parks would expand on this problem and its magnitude.

The characteristics of Lassen Volcanic NP's action-response region are likely to change following further analysis of this and other park fires. It is unlikely that this zone will ever be under the complete control of the park. Rather, it will continue to be heavily influenced by the priorities of adjacent jurisdictions. This zone is not required to lie entirely outside the park, but should fit as closely as possible to the administrative boundary in order to give the park the maximum amount of habitat under the influence of the park's own policies.

Zones similar to this one should be identified and incorporated into the general planning process of the administrative boundary region of national parks. This would be useful in interpreting how conflicts in management objectives, such as with fire, influence landscape structure and ecosystem processes. In addition, these flexible buffers should be used for conflict resolution and cooperative ecosystem management.

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The Giant Sequoia Fire Controversy: The Role of Science in Natural Ecosystem Management

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Abstract. Efforts to preserve, or where necessary restore, natural ecosystems in the national parks of the Sierra Nevada have focused attention on questions of policy, science, and practicality. Management efforts to restore fire as a natural process in the giant sequoia (*Sequoiadendron giganteum*) groves of Sequoia, Kings Canyon and Yosemite National Parks provide a clear example of such questions. Following challenges regarding the scientific rationale, management objectives and visual impacts of prescribed burning among these giant monarchs, the National Park Service has undertaken a comprehensive research program to better understand the history and role of both fire and fire suppression in sequoia-mixed conifer forest communities. Individual projects address questions of fire and vegetation history, forest structure and dynamics, fire and fuel modeling, fire effects, and visitor perception. Research results should provide a basis for refining and articulating objectives, as well as for restoring fire to as natural a process as possible.

INTRODUCTION

The preservation of magnificent groves of giant sequoia (*Sequoiadendron giganteum*) was one of the driving forces behind the creation of Sequoia National Park. Today, following nearly a century of effective fire suppression in an ecosystem thought to be adapted to a fire interval between about ten and thirty years (Kilgore and Taylor, 1979), the sequoia groves may be threatened. In changing the historic disturbance regime through fire suppression, the National Park Service (NPS) has changed the forest. These changes are best typified by a buildup of ground fuels and a shift in relative species composition. Prescribed fire is currently being used as a tool to reduce unnatural fuels and restore fire as a natural ecological process. Recent challenges, both to the magnitude and importance of changes caused by a century of suppression and the acceptability of the visual impacts following fire, have pointed to areas in need of additional investigation. A comprehensive research program is underway to better understand fire and vegetation history, as well as the effects of variable intensities and frequencies of fire on the sequoia ecosystem.

EARLY HISTORY

Since the discovery of giant sequoia in isolated groves of the central and southern Sierra Nevada of California (Rundel, 1972), efforts have been made to capitalize on these large trees. Logging operations soon devastated entire groves (Hartesveldt et al., 1975). Fortunately, while the inaccessibility of the groves and poor wood quality made the trees low in value for most commercial purposes, other interests recognized the unique qualities of these giant trees and soon pushed for their protection. Efforts by local conservationists led to the permanent protection of some of the prime uncut sequoia lands (Hartesveldt et al., 1975).

In the decades following the 1890 creation of Sequoia, General Grant (later incorporated into Kings Canyon) and Yosemite National Parks, national park management emphasized the protection of scenery and natural objects (Bratton, 1985; Graber, 1983). During this era of protection, naturally ignited fires, which were thought to "hurt" the trees and cause visual blemishes (black bark and scorched foliage) were suppressed (Tweed, 1987). Through the early 1960s no significant fires, human- or lightning-caused, had burned in any of the sequoia groves of the three parks.

In a 1963 report to the Secretary of Interior, a Special Advisory Commission on National Parks warned that continued fire suppression would cause an increase in hazardous fuels and change plant and animal communities in the conifer forests of the Sierra Nevada (Leopold et al., 1963). The Leopold Commission called for active management to restore and maintain more natural ecosystems. They further recognized the importance of historical and ecological research, including small-scale experimentation, to better understand biotic relationships and the impacts of human activities on park ecosystems.

The Role of Fire in Sequoia Forests

Studies carried out in the 1960s and early 1970s clearly documented the importance of periodic fire in maintaining the giant sequoia mixed-conifer forests of the Sierra Nevada. For millennia fire has reduced flammable surface fuels, thinned forest trees, stimulated sprouting of shrubs and other hardwoods, released seeds and prepared seedbeds favorable for germination of giant sequoia, efficiently recycled nutrients, and influenced insect and disease populations (Harvey et al., 1980; Kilgore, 1972). The implementation of fire suppression policies disrupted these relationships

(Kilgore, 1973). Since much of our understanding of sequoia ecology is based on limited research, a full quantitative understanding of these impacts is still largely lacking. Nevertheless, by the late 1960s the National Park Service had begun a program of prescribed burning in the sequoia groves. The goal was to reduce the hazardous fuel buildup in the hope of avoiding a holocaustic wildfire, while at the same time restoring fire to a more natural role (Bancroft et al., 1985).

Prescribed Fire Management

Without a clear understanding of subtle ecological interactions or a well defined goal, early burning in Sequoia and Kings Canyon National Parks focused on the narrow objectives of using strip headfires of uniform intensity to remove up to 100% of the down fuel and shade-tolerant understory trees. Such burning ignored the patchiness believed to have characterized pre-settlement fires. It was obvious that if the goal was to perpetuate natural ecosystems, a way had to be found to allow fire to play a more natural role (Parsons and Nichols, 1986).

The 1980s saw an increased sensitivity to the variability of the natural fire regime. Spot fires, backing fires, and night burning were all used to increase the heterogeneity of fire effects. Objectives were refined to include the breaking up of homogeneous fuel accumulations to restore conditions thought to be within the range of natural variability. It was hoped that someday natural ignitions might be allowed to burn in a manner they would have had modern man not intervened (Parsons et al., 1986). It was recognized that where concerns of safety, health, facilities, smoke, or other human or resource values would not permit natural ignitions, prescribed burning would continue to be used in the place of natural fire. These refinements in goals and objectives were accompanied by an ever-increasing sophistication in fire science. These years were also characterized by an increasing recognition of the importance of understanding natural fire regimes and the effects of variable fire frequencies and intensities on ecosystem properties. Questions as to whether, when, how often and how hot to burn a given area had to be answered. A decision was made in the summer of 1985 that an outside review was needed to focus on the information needs of the prescribed fire program.

QUESTIONS RAISED

Questions in need of review included the magnitude of change in forest structure and fuel accumulation that had occurred as a result of fire suppression, as well as an understanding of past variability in fire history, climate and forest dynamics (Bonnicksen and Stone, 1982, 1985). In the fall of 1985, additional questions were raised regarding aesthetic or visual resource impacts of prescribed fire in the sequoia groves (Cotton and

McBride, 1987). Specifically, concern was expressed over the bark char, foliage scorch and occasional fire scar enlargement resulting from the park's prescribed fire program. Cries of "sequoias in flames," "scenery crisis," "administrative despotism" and "professional vandalism" were raised by a former local resident concerned with the park's fire management program (E. Barnes, unpublished correspondence). Conservation organizations such as the Sierra Club, Save the Redwoods League and the National Parks and Conservation Association expressed interest and concern over the issues raised.

Outside Review

In the spring of 1986, NPS Western Regional Director Howard Chapman appointed a seven-member, independent panel of outside experts to "review the prescribed fire program in the sequoia-mixed conifer forests at Sequoia, Kings Canyon and Yosemite National Parks." Dr. Norm Christensen, Professor of Botany at Duke University and Chairman of the Vegetation Section of the Ecological Society of America, served as panel chairman. Other members included Dr. Joe McBride (Forestry and Landscape Architecture, UC Berkeley), Dr. Thomas Harvey (Botany, San Jose State Univ.), Dr. Robert Martin (Forestry and Conservation, UC Berkeley), Dr. Ronald Wakimoto (Forestry, Univ. Montana), Dr. Philip Rundel (Environmental Biology, UCLA), and Lin Cotton (private consultant in landscape architecture). Panel members were selected to include individuals with extensive experience with fire ecology, fire management, or the ecology of sequoia-mixed conifer forests, but who had not been specifically involved with any aspects of the sequoia fire "controversy." Individuals and groups concerned with the issues to be reviewed were invited to address the panel. The panel met and acted independently of the National Park Service.

The panel was charged with reviewing the history, current status and scientific basis for the sequoia-mixed conifer fire management program. It was asked to evaluate the impacts of prescribed as opposed to natural fire on the giant sequoia and to review the specific fire management objectives and the means employed to judge relative success or failure. The panel met for three days in July, 1986 at Sequoia National Park. The meeting included presentations on the fire program by Park Service managers and scientists, as well as comments by concerned individuals and groups. It included a full-day field trip to examine recent burns of varying intensity, unburned areas, and areas proposed to be burned in the near future. Numerous follow-up meetings and exchanges of information and ideas by the panel eventually led to a final report and formal presentation of recommendations to NPS Director William Penn Mott, Jr. in February, 1987 (Christensen et al., 1987).

PANEL RECOMMENDATIONS

In addition to reviewing management policy and its implementation, the panel report focused on specific research needs for the sequoia-mixed conifer ecosystem. Recommendations included the need for in-house and collaborative research on fire history, demographic and life history studies, fuel dynamics, computer simulation models, visitor response, and fire effects (Christensen et al., 1987). It was felt that information of this kind was essential to making critical management decisions. While previous studies had documented the unnaturalness of fire suppression, they had provided little of the detail necessary to fully understand either the impacts of suppression or the outcomes of alternative management scenarios designed to either restore ecosystem processes or reconstruct forest structure. Furthermore, the lack of detailed understanding of basic ecosystem structure and function limits the use of quantitative standards so necessary for evaluating the success or failure of a management program (Bonnicksen, 1983). While many of these needs had been previously identified, a general perception had existed that the Sierra Nevada parks had model fire management programs, based on sound scientific rationale. Internal appeals for additional funds to carry out needed studies had received little attention. It took the perception of a crisis to again place these needs in a competitive position.

The Research Program

The months following issuance of the Christensen Report, and subsequent attention given to its recommendations (e.g., Challacombe, 1987), saw a rallying of the funds and scientific expertise necessary to address many of the most important questions. A combination of funds from the NPS' Washington Office Natural Resources Preservation Program, the Western Regional Office Natural Science Program (WR), the Yosemite and Sequoia-Kings Canyon fee legislation and park base (YOSE and SEKI), the U. S. Forest Service (FS), and private (Sequoia Natural History Association; SNHA) funds have permitted implementation of a comprehensive research program. Coordinated jointly between, and carried out within, both Yosemite and Sequoia-Kings Canyon, the program combines NPS, university and Forest Service scientists in a multidisciplinary study of fire and vegetation history, forest and fuel dynamics, fire effects on vegetation, soil and pathogens, and visitor perception of the fire management program. Specific study components (listed in Table 1) include the following:

Biogeography and disturbance history of Sequoiadendron. Recent evidence of significant changes in forest composition and charcoal deposition in the central and southern Sierra Nevada during the past 10,000 years (Davis et al., 1985) has raised questions as to the uniqueness of the recent era of fire suppression. Dr. R. Scott Anderson of Northern Arizona University is examining pollen, plant macrofossils and charcoal (through thin section analysis) from sediment cores in meadows within or adjacent to modern sequoia groves in an effort to understand environmental conditions and vegetation composition through recent millennia. Such information is critical to quantifying the divergence of current conditions from natural variation in forest structure and fire frequency. For example, should the sequoia-mixed conifer forest type be shown to have been continually changing in response to recent changes in climate and fire frequency, the argument could be made that a century of fire suppression and associated shifts in forest structure might be less important than similar changes in a more homeostatic system. An understanding of the variance of both ecosystem properties and disturbance regime provides critical input to the tough decisions that must be made regarding the extent and type of ecosystem restoration required in a given area. The extraction of sediment cores dating approximately 10,000 years before present from several sequoia zone meadows, together with preliminary indications of significant changes in climate and vegetation and a long record of periodic fire, give considerable hope for a much improved understanding of past environmental conditions.

Tree-ring reconstruction of giant sequoia fire history. A critical element in restoring a semblance of natural fire regimes is an understanding of past fire frequency and intensity. Decisions as to when and how often an area should be burned depend, in part, on what is known about past fire occurrence. Because of the longevity of individual giant sequoia, a time perspective spanning many centuries and even millennia is required to fully understand the role of fire in the ecosystem. Research directed by Dr. Tom Swetnam of the Laboratory of Tree Ring Research at the University of Arizona utilizes fire scars on sequoia stumps and logs to reconstruct the frequency and extent of fires over the past one to two thousand years. The well preserved wood and clear fire scars have permitted, with the use of crossdating, the accurate dating of dozens of scars on individual specimens. By noting the position of scars within individual rings, the reason of individual fires can also be inferred. In addition to indications that the older trees may have "experienced more than 100 surface fires," it is interesting to note that the oldest fire scar observed to date occurred in the year 931 B.C. (Swetnam, 1988). Fire scar records are being obtained for at least five separate sequoia groves and multiple sites within at least one grove. Such accurate reconstructions of past fire frequency, seasonality and intensity will provide a major tool for future fire management planning.

Fire and forest dynamics of giant sequoias. Despite a general understanding that fire in sequoia groves often results in successful establishment of giant sequoia seedlings, no data exist regarding conditions necessary for sequoia recruitment. For example, is fire alone, regardless of intensity or frequency, enough to ensure both the establishment of seedlings and their subsequent recruitment as mature sequoias? How do fire intensity, severity, season of burning, and climate interact in determining successful recruitment? To address these questions, the Sequoia-Kings Canyon research office (Drs. Nate Stephenson and David Parsons) has undertaken a study of forest structure and composition. Analysis of the age structure of selected stands of giant sequoia, for which 1000 years or more of fire history will soon be available, should help establish the link between fire and sequoia recruitment, including an understanding of the nature of past fires that led to such recruitment. Preliminary findings show that areas with dozens of recorded fire scars may contain a much more limited number of tree age classes, indicating that successful sequoia recruitment depends on a combination of factors.

Fire and fuel dynamics in sequoia-mixed conifer forests. Predictions of fire behavior and effects are based largely on an understanding of forest fuel dynamics. Dr. Jan van Wagtendonk of Yosemite National Park is modeling fuel accumulation rates coupled with stand growth and fire behavior prediction models under varying fire regimes. Simulation of fuel buildup under a lightning ignition regime, or some known historical fire frequency, allows comparison of fuel loadings and predicted behavior under such conditions with those existing following varying periods of fire suppression. Such information will provide park managers with a basis for making decisions on when, where, how often, and how hot to burn, as well as predicting the effects of varying fire intervals and intensities on fuel loading and forest succession (van Wagtendonk, 1985).

Fire effects on forest pathogens. The interactions between fire and pathogens in sequoia forests are largely unknown, yet potentially significant. Among the myriad effects of fire and fire suppression on sequoia ecosystems, potential effects on pathogen populations have been identified for special attention (Christensen et al., 1987). Drs. Doug Piirto (Cal Poly University, San Luis Obispo) and J.R. Parmeter (University of California, Berkeley) have undertaken studies to: (1) better understand the host specialization and possible cross-infectivity of the common root fungi *Fomes annosus;* and (2) determine the effects of fire scars and their re-burning on the incidence, extent, and survival of decay fungi in giant sequoia. This information should provide an improved basis for evaluating the ecological impacts of alternative fire management scenarios.

Prescribed fire effects on forest fuels and soil and cambium temperatures. One of the most important effects of fire on the forest ecosystem involves the combustion of ground fuels. When litter, duff and woody fuels have been permitted to accumulate for long periods without fire, the subsequent combustion can result in elevated soil and bark temperatures and potential negative effects on root systems and cambial tissue. The Forest Service (Riverside Forest Fire Lab) is sponsoring a cooperative study directed by Steve Sackett and Sally Haase to investigate the effects of prescribed fire under varying prescriptions on ground fuel consumption and soil and cambium temperatures. Indications of potentially lethal temperatures will point to the need to burn under more limited prescriptions and/or carry out additional studies to examine potential impacts on the biological system.

Spatial and temporal variation of understory vegetation and environmental heterogeneity in giant sequoia-mixed conifer stands. A lack of quantitative information on conditions influencing seedling establishment and survival for tree and shrub species in the mixed conifer forest has limited efforts to accurately model forest stand dynamics and succession. Stephanie Gebauer of Duke University will undertake a study in 1989 to evaluate the spatial and temporal variation of understory vegetation in relation to such environmental factors as overstory cover, radiation, temperature, moisture, and soil nutrients. These data should greatly improve our understanding of the life history and demography of the giant sequoia and associated species.

Visitor perception of fire management program. The need to better understand the park visitor's response to and understanding of the fire management program has been addressed through a recent study by Dr. Joyce Quinn of California State University, Fresno. A survey of 1,000 visitors to Sequoia and Kings Canyon in 1987 (Quinn, 1988) indicated an awareness of the parks' prescribed fire program, including an understanding that the primary purpose of the program was to prevent uncontrolled wildfire. The survey also showed a strong motivation for park visitors to experience a "natural" environment; a feeling that added to the general support expressed for the idea of overcoming the unnatural effects of fire suppression.

OTHER STUDIES

In addition to the above major programs funded in direct response to the Christensen Panel recommendations, a number of other studies currently underway or planned for the near future promise to provide valuable information in improving our understanding of sequoia-mixed conifer fire regimes and their effects on ecosystem properties. These include the use of trace chemical analysis to detect possible chemical signals of past fires in tree rings (J. McBride, UC Berkeley); climate reconstruction from tree rings (L. Graumlich, U. Arizona); Native American use of fire (K. Roper-Wickstrom and K. Anderson); size structure and spatial pattern analysis of giant sequoia (T. Stohlgren, NPS-UC Davis); impacts of fire

TABLE 1. Sequoia-mixed conifer fire research projects underway in Sequoia, Kings Canyon and Yosemite National Parks.

	Study	Principal Investigator	Institution	Years	Funding Source*
1.	Biogeography and disturbance history of *Sequoiadendron*	R. Scott Anderson	No. Ariz. Univ.	87-90	SEKI/YOSE
2.	Tree-ring reconstruction of giant sequoia fire history	Tom Swetnam	Univ. Ariz	87-90	NRPP
3.	Fire and forest dynamics of giant sequoias	Nate Stephenson David Parsons	NPS/SEKI	87-91	WRO
4.	Fire and fuel dynamics in sequoia-mixed conifer forests	Jan van Wagtendonk	NPS/YOSE	87-91	WRO
5.	Fire effects on forest pathogens	Doug Piirto J. R. Parmeter	Cal Poly Univ. UC Berkeley	88-90	NRPP
6.	Prescribed fire effects on forest fuels, soil and cambium temperatures	Steve Sackett Sally Haase	USFS, Riverside	88-90	FS/SEKI
7.	Spatial and temporal variation of understory vegetation	Stephanie Gebauer	Duke Univ.	89-90	NRPP
8.	Visitor preception of fire management program	Joyce Quinn	CSU Fresno	87-88	SNHA

*SEKI/YOSE = Sequoia/Kings Canyon - Yosemite; NRPP = Washington Office Natural Resources Preservation Program; WRO = Western Regional Office; USFS = U.S. Forest Service; SNHA = Sequoia Natural History Association.

suppression on forest aggregations (T. Bonnicksen; Texas A and M); nitrogen cycling (P. Matson, NASA-Ames); soil solution chemistry (J. Chorover, UC Berkeley and P. Vitousek, Stanford); and ozone stress on conifers (P. Miller, N. Grulke, USFS and M. Patterson, UCLA). Efforts in the area of forest succession and stand growth modeling are also anticipated. Additional information such as birth and mortality rates, necessary for fully evaluating the fire management program, will be obtained from expanded fire effects monitoring.

CONCLUSIONS

The use of fire to restore or maintain naturally functioning biological communities requires a detailed understanding of ecological processes and their effect on the structure and function of those ecosystems. Like many disturbance phenomena, fire in the Sierra Nevada sequoia-mixed conifer forests has and will continue to occur at varying intervals and intensities. Understanding the spatial and temporal variance of this disturbance regime is critical to making future decisions as to whether, where, when, how often and how hot to burn a given area. The research currently underway in Sequoia, Kings Canyon and Yosemite National Parks on fire and vegetation history, and fire, fuel and forest dynamics will provide improved perspectives of the historic variance of these properties, the extent to which current conditions depart from that natural variability, and thus, the extent to which ecosystem restoration or reconstruction may be required (Bonnicksen, 1988). Such information will also provide quantifiable standards for evaluating the success or failure of the management program. Likewise, understanding the effects of variable intensity (including variation in season and severity) fires on such ecosystem properties as vegetation (germination, recruitment and mortality), fuels, and pathogens is essential to predicting the impacts of any given prescribed fire. The research on forest dynamics, understory vegetation, fuels, pathogens, and prescribed fire temperatures, together with the comprehensive fire effects monitoring programs underway in each park, should contribute greatly to our understanding of the effects of varying fire regimes.

A second basic dilemma facing park managers is weighing the importance of preserving and perpetuating the natural ecological processes that have determined the nature of the sequoia groves as opposed to controlling these processes to achieve some other state, such as a desired perception of the park (Cotton and McBride, 1987). If the natural systems that produced this unique forest conflict with our sense of aesthetics, we may need to reexamine the goals by which we manage these areas. Current NPS management policies for natural area parks emphasize the preservation of natural ecosystems. And while the definition of "natural" is open to considerable debate, there is little doubt that fire has long played an important role in the evolution of the sequoia-mixed conifer forest. It must again be permitted to do so if a semblance of a natural ecosystem is to be preserved.

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